# FEASIBILITY OF USING A RISK - BASED APPROACH TO SET INTEGRATED ENVIRONMENTAL OBJECTIVES FOR THE PROTECTION OF WATER RESOURCES

S Jooste • HM MacKay • P-A Scherman • WJ Muller

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## EXECUTIVE SUMMARY

The Department of Water Affairs and Forestry (DWAF) has embarked on a project to develop and implement a protection-based classification system for water resources, in order to meet the objectives of the National Water Policy (DWAF, 1997) and National Water Act (No. 36 of 1998). The following factors are inherent to the classification project and set the framework for the initiation of the risk-based objectives (RBO) project

- Not all water resources will have the same level of protection, with each Ecological Management Class (EMC) carrying specific levels of protection or levels of *risk* of damage to the sustainability of the ecosystem.
- Resource Quality Objectives (RQOs) 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.

As a result of these factors, it became obvious that methodologies needed to be developed to set measurable, verifiable integrated objectives for water resource protection. These methodologies need to incorporate factors inherent to biological systems and be contained within a risk-based approach, which would encompass the uncertainty and variability of biological data.

The general aim of this project was therefore to investigate the feasibility of using a risk-based approach to setting RQOs for the protection of water resources. 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.

Specific aims 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 RQOs for protecting water resources;
- produce a report outlining potential approaches for setting integrated RQOs for the protection of 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.

The report is presented in three parts, with Part 1 comprising background literature on risk concepts and the use of a risk-based approach in water resource management. Parts 2 and 3 summarise the findings of two specialist workshops which considered the feasibility of following a risk-based approach, and looked at flow as a specific stressor, respectively.

**Part 1** assesses the feasibility of using a risk-based approach to setting integrated environmental objectives for the protection (sustainability and resilience) of water resources, and consists of four chapters

*Chapter 1* details the background to RQOs and how they can be applied to water resource management, and introduces the concept of risk and how it can be applied and used. Concepts such as sustainability and resilience are introduced and defined, and their role in underpinning water resource protection is explained. The role of RQOs in the classification of water resources and the Ecological Reserve are also explained

*Chapter 2* focuses on the concept of risk and details its potential and practical usefulness in water resource managements, as well as its role in the classification system. This Chapter therefore looks at the integration of risk objectives and risk criteria (acceptable risk), with ecological and management objectives.

*Chapter 3* discusses the realistic feasibility of applying risk and risk-based objectives in setting RQOs for resource management, and provides a succinct summary of the specialist workshops (see Parts 2 and 3) which were held to assess the feasibility of using risk in water resource management.

Part 1 concludes with ***Chapter 4***, which identifies research needs and recommends a number of research directions, enabling the effective use of a risk-based approach to setting integrated environmental objectives. Subjects covered include the need for risk management structures and policy, and the importance of understanding risk concepts and improving its accessibility to practitioners and managers. Although stressor-response relationships are discussed in Parts 2 and 3 of this document, the integration of co-occurring stressors is 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, is also emphasized.

**Part 2** documents the discussion and findings of a workshop on the use of RBOs in water resource management, and highlights the requirement for information on stressor-response relationships.

One of the key features of a risk assessment, particularly in characterizing and evaluating the probability of effect, is information on *stressor-response relationships*. 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.

Stressor-response relationships were therefore reviewed and discussed at a specialist workshop in August 1998. Literature was reviewed for information on functional relationships which may 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, EC, salinity, TDS, TSS, temperature
- habitat

Discussions around stressor-response relationships highlighted the dearth of information on functional relationships. It was recommended that instead of focussing resources on the development of a large toxicity database for a range of South African organisms, South African data currently available should be utilized and compared to international data. If toxicity information is similar, international data can be used for the present, and more local data be included as it becomes available. Mesocosm population testing may need to be undertaken, and mathematical models used to extrapolate results to field conditions. Laboratory-based toxicity data can also be integrated with observed effects in the field (e.g. biomonitoring data). For the moment, there should be a reliance on expert opinion, particularly for parameters such as flow.

**Part 3** documents the discussion and findings of a workshop on using risk-based objectives to set flow requirements for rivers. This workshop was held as a result of the outcome of the RBO workshop held during 1998. Two different methods were tested for setting the quantity component of the Ecological Reserve, and for altering flow requirements for various Ecological Management Classes (EMCs). These two methods were as follows:

- The '*less frequency/assurance*' method, which generates different assurances of maintenance flows for different EMCs, i.e. maintain the depth, velocity and wetted perimeter of the maintenance flow, but alter its assurance or frequency.
- The '*less depth*' method, which motivates for higher/lower flows for different EMCs, i.e. a change in maintenance flow (volume) per EMC.

The two different methods were applied to rivers with a range of hydrological variability for which Instream Flow Requirement (IFR) data was available, and the hydraulic and ecological consequences of applying each method assessed. The results did not show a generic preference for either method when setting maintenance flows for different EMCs. It is possible that each case should be handled independently, the suitability of each method assessed, and the appropriate method selected. Although useful, the assessment of both methods would be costly and impractical. It was suggested that a decision framework be developed which could guide scientists in their assessments.

Documents written by Hughes and O'Keeffe respectively, are also included in Part 3, and are 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.

Hughes ***(Part 3, Document 1)*** attempts to quantify the ecological risk associated with adopting different EMCs (or different proportions of the natural flow regime), by defining approximate stress/flow relationships for individual components of the system (e.g. fish, invertebrates, riparian vegetation) in the form of stress curves generated by biologists. Once a stress level can be correlated with any discharge, hydrological time series can easily be converted to stress-level time series, and analysed for stress frequency/magnitude characteristics. Any historical, present or potential flow regime can then be compared with any other, to assess the resulting stress profile.

O'Keeffe ***(Part 3, Document 2)*** attempts to define a qualitative description of stress levels which could be used to construct stress curve relationships. His example refers to instream fauna which require flowing water for optimal habitat conditions. Stress is consequently seen in terms of reduced flows and loss of hydraulic habitat. The time dimension of stress is taken into consideration when the stress curves are related to hydrological time series to define stress profiles. A stress index of stress levels 1 to 10 is presented as an example, where stress level 1 describes a condition of no stress, with extensive availability of critical biotopes. Stress level 5 shows a moderate loss of abundance of critical hydraulic habitats, where species with flow-dependent breeding habits will not breed, but should survive. Stress level 10 is the absence of all surface water, and loss of species other than those with specialist survival behaviour due to lack of hydraulic habitat. The index is intended to be used to describe flow/stress relationships in specified river reaches, from which stress profiles can be generated (as described above).

## GLOSSARY

| | |
|---|---|
| **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 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). |
| **Ecological risk assessment** | The process of defining and quantifying risk to nonhuman biota and determining the acceptability of those risks (Suter, 1993). |
| **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 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, in press).

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

**Resource** "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 (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 instream flow;
(b) the water quality, including the physical, chemical and biological characteristics of the water;
(c) the character and condition of the instream 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 (Part 1, Section 1.2.4).

**Risk** The probability of a prescribed undesired effect. If the level of effect is treated as a number, risk is the product of the probability 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 magnitudes and probabilities to the adverse effects of human activities or natural catastrophes (Suter, 1993).

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| **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 behavior may be analysed statistically but not predicted precisely (Brown, 1993). |
| **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** | 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;<br>(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 ABBREVIATIONS

| | |
|---|---|
| **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 |
| **CV** | Criterion Value |
| **DSS** | Decision support system |
| **DWAF** | Department of Water Affairs and Forestry |
| **EC** | Environmental Concentration |
| **EMC** | Ecological Management Class |
| **EPA** | (United States) Environmental Protection Agency |
| **ERA** | Ecological Risk Assessment |
| **ERBM** | Ecological Risk-Based 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 |
| **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 |
| **TWQR** | Target Water Quality Range |

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# PART 1

# THE FEASIBILITY OF USING A RISK-BASED APPROACH TO SET INTEGRATED ENVIRONMENTAL OBJECTIVES FOR THE PROTECTION OF WATER RESOURCES

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## 1.1 INTRODUCTION

### 1.1.1 The potential value of a risk-based approach

There are two driving forces that have led to the need to assess the feasibility of following a risk-based approach in setting **Resource Quality Objectives** for water resources. These are:

- firstly, the need to balance long-term protection of water resources with their short-term utilisation, while promoting equity, access to basic water services, and economic development; and
- secondly, the need for an approach to water resource management which can take into account the complexity of ecosystem processes, and their inherent uncertainties.

South Africa is primarily a developing country. The Water Policy (DWAF, 1997) and the National Water Act (NWA; No. 36 of 1998) recognise the need to utilise water resources for optimal social and economic benefit in the short term, but within the constraints of what is environmentally sustainable in the long term. It is understood that not all **water resources** can, or should, be given the same level of protection. In some cases, the urgency of short term demands for services and development may lead to acceptance of a higher risk of irreversible damage to a water resource; in others, the importance and sensitivity of a particular water resource may be such that only a very low risk of damage is acceptable, and the impacts of short term demands would be limited accordingly. These different levels of protection are discussed in more detail in a later section.

Recently much world-wide research attention has focused on the second driving force, namely development of management tools which can be used in the context of the inherently uncertain nature of ecosystems and their processes. If management tools are to achieve their goal, then resource management objectives and associated numerical or narrative criteria, assessment methods and management processes must reflect inherent ecosystem characteristics. The ecosystem characteristics are defined by the dynamics and kinetics of interactions between aquatic animals, plants, habitats and processes that determine the function, composition and diversity of the aquatic ecosystem at various levels of organisation, from landscape to molecular level (Noss, 1990).

Identifying the two driving forces led to the need to develop a suitable approach for setting Resource Quality Objectives. A number of fundamentally different approaches for setting Resource Quality Objectives are available:

- a technology-based approach, which emphasises the current state of technology with or without reference to the economic feasibility (e.g. best available technology BAT, BATNEEC, BPT etc.)
- an economics-based approach that emphasises the cost and possibly socio-economic impacts
- an effects-based approach that emphasises the impacts without necessarily referring to

technology or economics. Effect-based approaches include:

a) the **hazard-based** approach which emphasises the potential for causing an effect and

b) the **risk-based** approach which emphasises a realistic expectation of effect.

The risk-based approach has been proposed as one which will allow the importance and value of resources to be recognised within the structured national framework of a water resource management classification system, with classes representing different levels of risk of irreversible damage, ranging from high to low. It allows for the derivation of site-specific Resource Quality Objectives, which translates the level of risk associated with a selected management class into measurable, verifiable criteria for the resource.

Although the risk-based approach is scientifically more challenging and technically more demanding for resource managers, there was clearly a need to investigate developments in the fields of Ecological Risk Assessment and management, to determine whether these could be designed into a practical, usable framework to support South Africa's water policy.

### 1.1.2 Objectives of this project

The general aim of this project is to investigate the feasibility of using a risk-based approach to setting Resource Quality Objectives for protection of water resources. The current investigation is intended to provide direction for future, more detailed, research in order to support, develop and implement the proposed protection-based classification system for water resources, as outlined in the White Paper on National Water Policy (DWAF, 1997).

Specific aims are 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;
- produce a report outlining potential approaches for setting integrated Resource Quality Objectives for protection of water resources;
- develop a detailed research proposal for submission to the Water Research Commission in 1999. The proposal will address 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.

## 1.2 THE POLICY CONTEXT FOR RISK-BASED RESOURCE QUALITY OBJECTIVES

### 1.2.1 Water as a renewable resource

Some of the services provided by water resources, encompassed in the definition of utilisation, are:

- water for drinking and basic human needs;
- water for economic development (e.g. industry, agriculture, power generation);
- transport and/or purification of some waste products;
- subsistence or commercial supplies of fish and plants;
- opportunities for recreation;
- maintenance of habitats for conservation of particular plants, animals, landscapes and environments, and for conservation of biodiversity;
- opportunities for ecotourism;
- retention and storage of water;
- transport of flood waters.

Almost all of the services provided by water resources rely on natural hydrological, chemcial, biological and ecological processes, and require at least some degree of maintenance of the natural structure and character of the resource. As **"renewable natural resources"**, water resources have a certain amount of **resilience** to the pressures and demands of utilisation. This resilience allows water resources to be utilised on a continuous basis, as long as the demands are not too great.

If a water resource is over-utilised or allowed to degrade too far (*i.e.* too much water is taken out, too much waste is put in, natural shape and structure are modified too greatly by erosion, sedimentation or habitat degradation), the aquatic ecosystem loses resilience and begins to break down. The ecological integrity of the resource can be damaged: once this happens, the capability of the resource to meet people's demands for utilisation can be reduced, or possibly even lost altogether. If utilisation of water resources remains at a level within the ecosystem limits, to protect ecological resilience, then that level of utilisation can be sustained indefinitely.

In South Africa, many water resources have already been modified by utilisation and development, and no longer remain in their natural ecological state. However, a water resource does not have to be in a pristine or untouched state to have **ecological integrity**; even modified aquatic ecosystems can have resilience making them renewable resources, as long as they are managed in such a way that a certain level of ecological function and integrity is either maintained or rehabilitated.

### 1.2.2 A new perspective on water resources

The fact that water is now defined as a renewable natural resource (NWA, No. 36 of 1998) means a change in the way water resources are perceived and valued. This new perspective has two major implications for **water resource management**. Firstly, a water resource is not just water as the commodity but an entire ecosystem of which water is one component. The ecological integrity, which gives a water resource its resilience, is an essential component of the value of that resource. This leads to a new and broader definition of a water resource:

> *A water resource is an ecosystem which includes the physical or structural aquatic habitats (both in-stream and riparian), the water, the aquatic biota, and the physical, chemical and ecological processes which link habitats, water and biota.*

Secondly, the limits of utilisation which can be sustained by a water resource before resilience is lost must be recognised and respected. Resilience depends on maintaining a base level of ecological integrity and function. This level, called the **Resource Base**, is crucial to the capability of water resources to sustain utilisation, and must be protected.

In recognition of the importance of the Resource Base, the Water Law Principles (DWAF, 1997) identify a **"Reserve"**, which gives effect to the policy for protection. The Reserve is intended to protect the resilience of water resources, in order that basic human needs can be met (e.g. human health and safety and domestic water supply) and ecological functions and processes can be sustained. The Reserve is defined in terms of the quality of water as well as the quantity and assurance of water. Both of these are needed to provide basic human needs and protect the structure and function of ecosystems while ensuring ecologically sustainable development and utilisation.

### 1.2.3 Finding the balance

The responsible management of water resources includes a responsibility to protect the users of water resources, which in turn requires **protection** of water resources from over-utilisation or other impacts which cause degradation.

Protection encompasses:

- protection to ensure sufficient water quantity and water quality (especially in relation to human health), to meet basic human needs;
- protection of ecosystem structure and function, to ensure that utilisation of water resources can be sustained in the long term;
- meeting the water quality requirements of other water users (agriculture, industry, recreation) as far as possible, within the constraints of requirements for protection of basic human needs and protection of the resources.

**Sustainable utilisation** of water resources requires a balance between an acceptable level of long term protection of water resources, and society's present requirements for economic growth and development. For example, the total prevention of pollution, while an ideal for which to strive in the long term, is not practical in the short to medium term, since neither the emission of waste to the water environment, nor the impacts of land uses on the water environment, can be entirely prevented. However, they can, and must, be managed and regulated to achieve adequate long-term protection of water resource quality. This is the aim of the **resource protection policy**.

Implementation of the resource protection policy rests on the combined use of four types of regulatory activities:

- **resource-directed measures**, *i.e.* defining a desired level of protection for a water resource. Clear numerical or descriptive goals for the resource quality of the resource can be set (*i.e.* the Resource Quality Objectives);
- ***source-directed controls***, *i.e. controlling impacts on the water resource through the use* of i.) regulatory measures such as registration, licenses, directives and prosecution, and ii.) economic incentives such as levies and fees, to ensure that the Resource Quality Objectives are met;
- managing demands on water resources in order to keep utilisation within the limits required for protection;

- monitoring the status of water resources continually to ensure that the Resource Quality Objectives are being met, and to modify programmes for resource management and impact control as and when necessary.

### 1.2.4 Resource Quality Objectives

The Resource Quality Objectives for a water resource are a numerical or descriptive statement of the conditions which should be met in the receiving resource, to ensure that the resource is protected. Because they are a statement of resource quality, and not only water quality, the Objectives have four critical components which cover each aspect of ecological integrity necessary for protection of the Resource Base:

- Requirements for **water quantity**, stated as In-stream Flow Requirements (IFRs) for a river reach or estuary, or water level requirements for standing water or groundwater. IFRs are determined according to a set procedure;
- Requirements for **water quality** determined by current guidelines and procedures set out in the South African Water Quality Guidelines;
- Requirements for **habitat integrity**, encompassing the physical structure of in-stream and riparian habitats, as well as the vegetation aspects;
- Requirements for **biotic integrity**, which reflect the health, community structure and distribution of aquatic biota.

Resource Quality Objectives for a water resource are set on the basis of levels of acceptable **risk**: *i.e.*, in accepting a smaller risk of damaging the Resource Base, and possibly losing the services provided by the water resource, the more stringent the objectives would be. A higher risk to the Resource Base might be accepted in return for greater short term utilisation, in which case the Resource Quality Objectives would be set at less stringent levels.

### 1.2.5 The purpose and application of Resource Quality Objectives

Resource Quality Objectives set for a resource serve as the basis for water resource management. These objectives have a number of purposes in the context of resource management:

- Objectives are defined goals for the desired protection level and water resource quality status for management purposed. The goal may be achieved in steps as part of a long term programme with interim objectives used to define the steps and the time frame for achievement.
- Objectives provide a clear distinction between which activities and impacts are acceptable and those which are not. This includes the impacts of point sources, non-point sources, incident sources, land-use and development, and water abstraction.
- Objectives provide a quantifiable, verifiable baseline for measuring the success of resource management activities and for reviewing the effectiveness of source-directed control and regulatory activities.
- Objectives provide a stable framework, for an agreed time period, such that both resource managers and the regulated community can undertake decision making and planning.

## 1.3 SUSTAINABLE MANAGEMENT OF WATER RESOURCES

The Water Law Principles (DWAF, 1997) include several references to sustainability: that "human use of water [should] not...compromise the long-term sustainability of aquatic and associated ecosystems"; that "development, apportionment and management of water resources should be carried out using the criteria of public interest, sustainability, equity and efficiency of use"; and that all citizens have a right to "have access to the basic water services...necessary to afford them a healthy environment on an equitable and economically and environmentally sustainable basis".

Implicit in these Principles is the idea of sustainable management of water resources in South Africa. The role of the Department of Water Affairs and Forestry (DWAF) as custodian of South Africa's water resources is one of stewardship with a responsibility to consider not only the welfare and needs of the current generation, but also those of future generations. These principles are embodied in the philosophy of sustainability. The implications of a commitment to sustainable management are two crucial and separate but interdependent functions of water resources management:

- utilisation of the resources, in an efficient and effective manner, for the benefit of this and future generations; and
- protection of the resources, to ensure their ability to support utilisation for the benefit of this and future generations.

### 1.3.1 What is sustainability?

Robert Solow (1993) discussed the meaning of **sustainability** in respect of policy development. He indicated that it was not feasible to expect to hand over natural resources to future generations in exactly the same state of "pristineness" in which they are found, since this would preclude development or utilisation by the current generation. Not allowing utilisation of resources for the wellbeing of this generation would not allow us to address the problem of a lack of equity in this generation, although it might allow us to meet our commitment to inter-generational equity. "Sustainability is...an obligation to conduct ourselves so that we leave to the future the option or capacity to be as well off as we are... ...In making policy decisions we can take advantage of the principle of substitutability, remembering that what we are obligated to leave behind is a generalised capacity to create well-being, not any particular thing or any particular natural resource".

The philosophy of sustainability is broad and all encompassing and no single definition can adequately and consistently reflect people's interpretation of the philosophy. However, it is possible to define sustainable activities more clearly and rigorously, which aids understanding of what the philosophy of sustainability entails. Bidwell (1993) describes "sustainable activities" as those activities which:

- are viable in the long term, *viz.* they do not reduce the long term possibility of meeting human needs; and
- preserve quality in the long term, where quality could be heritage, ecology, culture, diversity or productivity.

New Zealand's revised environmental legislation defined sustainable management as:

"Managing the use, development and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic and cultural well-being and for their health and safety while -

a.) sustaining the potential of natural and physical resources (excluding minerals) to meet the reasonably foreseeable needs of future generations; and
b.) safeguarding the life-supporting capacity of air, water, soil and ecosystems; and
c.) avoiding, remedying or mitigating any adverse effects of activities on the environment".

Each definition of sustainability reflects a particular point of view. However, in relation to renewable natural resources, such as water, there is a common thread through most definitions. Defining sustainability for renewable natural resources usually, in some way, involves maintenance of the diversity, complexity and function of these resources to maintain their resilience to disturbance and exploitation.

### 1.3.2 The concept of resilience

At a functional level, sustainability has been defined as "the ability of an ecosystem to support itself despite a continued harvest, removal, or loss of some sort" (EPA, 1997). Related to sustainability is the resilience of the resource, which refers to the ability of the system to adapt to change or stress. Both sustainability and resilience will be influenced by biodiversity. Biodiversity is composed of three primary attributes: composition, structure and function, and each of these may describe various levels of organisation, from landscape to molecular level (Noss, 1990). The maintenance of biodiversity, complexity and function of renewable natural resources is closely related to, and essential to, the maintenance of resilience. Evidence shows that loss of resilience, as a result of over-exploitation, causes an ecosystem to change, possibly irreversibly, to another ecological status (Arrow *et al.*, 1995; Walters, 1986). Such an ecosystem change can mean:

- loss of ecological functions, and therefore reduced ability to meet human demands, or even support human life;
- loss of the ability of the ecosystem to recover from natural disturbances such as floods and droughts;
- loss of options for future generations (e.g. depletion or contamination of groundwater reservoirs; soil erosion; salinisation); or
- increasingly expensive technologies will be needed in order to still meet the demands placed on the resource as it was in its previous ecological status (e.g. treatment of water; inter-basin transfers; specialised land management practices).

Resilience does not necessarily mean the ability of a renewable natural resource to return to its original unperturbed state once the pressure of the demand has been removed. Rather it is the ability of the resource to return to an ecological state that can meet the same or equivalent demands on it in the future. Measures of resilience are usually integrated measures, or indices, of aspects such as biodiversity, integrity, productivity and quality. Therefore the resource does not have to, and is highly unlikely ever to, actually *look* the same after the demands are removed, but it does have to have the same ability it once possessed.

The degree to which a resource should "look the same" is a reflection of its intrinsic value. Thus, if a resource has a very high intrinsic value, risks to changing its outward appearance by allowing any exploitation, even though it may be possible to restore biodiversity, integrity and quality once exploitation has ceased, are undesirable. Special management of human activities and impacts will be required to maintain the current outward appearance of a resource with a high intrinsic value but already impacted by human activities, for example, the Sabie River in the Kruger National Park.

The resilience of a resource, rather than its outward appearance, is the "natural capital" which must be protected, in order that this generation's custodians may hand over the undiminished capital to the next generation.

### 1.3.3 Protecting resilience

In section 1.3.2 it was suggested that there is a "point of no return", when natural resources are exploited beyond a certain level and resilience is irretrievably lost. The idea that there is a discrete threshold at which a resource changes to a new ecological state, within a narrow band of change and response, is artificial. This is because ecosystems, and especially aquatic ecosystems, are highly dynamic and in southern Africa especially aquatic ecosystems have adapted to a high natural variability. Although pressures of demand and exploitation can lead to irreversible changes in ecosystem state, the exact level (the "point of no return") at which an irreversible change will occur can be difficult to quantify. In part, this is due to the resilience of these ecosystems.

Irreversible change in ecological status can mean loss of resilience. Loss of resilience implies unsustainable management, which would be in conflict with the philosophy of sustainability. So how can resilience be protected?

It may not be possible to identify fixed levels of exploitation (water quantity and water quality) at which resilience is either lost or protected. However, based on measurements and understanding of the responses of ecosystems and laboratory systems, and especially their response under stress, a Resource Quality Objective which represents **an acceptable level of risk of approaching the "point of no return"** can be quantified. The higher the intrinsic value of the resource, the smaller the risk of change which can be accepted, and so the Resource Quality Objectives would be increasingly stringent. This is the approach used in the toxicity-based water quality criteria for aquatic ecosystems (DWAF, 1996), and, to some extent, in setting minimum ecological flow requirements (IFRs).

### 1.3.4 The concept of the Resource Base

Water resources have the capacity to meet human needs, within certain limits, and still be resilient. This means that whatever ecological "capital" is required to maintain resilience must be protected, but there could still be some capacity to meet additional demands.

The assurance that demands can be met on a sustainable basis rests on the assurance that the ecological resilience of the water resource is protected. The resource capacity should not be exploited beyond the level at which resilience is lost. If that level is exceeded, there is a risk of irreversible damage and subsequent loss of capacity to meet human needs in future. It is *important to note that it is loss of resilience that is to be prevented, not necessarily change in* outward appearance. For the purposes of development of policy and legislation this level or "point of no return" is called the Resource Base.

The Resource Base is defined, in conceptual terms, as "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".

It is worth noting that there are often several alternative ways to meet a variety of human needs from the same resource, whether those needs are, for example, for economic development, basic domestic water supply or recreational opportunities. Each of these alternatives might have a greater or lesser impact on resilience. An important aspect of the philosophy of sustainability is that any sustainable management process must allow the search for, and evaluation of, alternatives to meet human needs.

### 1.3.5 Resource Quality Objectives and the Ecological Reserve

In the Water Law Principles (DWAF, 1997), the Reserve is specifically identified as that water quantity and quality necessary to protect basic human needs and aquatic ecosystems. The Resource Quality Objectives for a water resource are a rigorous numeric or descriptive statement of the requirements of the Reserve for that particular water resource, in measurable, enforceable terms.

It is especially important to note that the Reserve is **not just the minimum** water quantity, water quality, habitat and biotic integrity required for protection of the resource. For a water resource classified with a high protection class, the Reserve would be set at a higher level, which corresponds to minimum risk and maximum caution. For a water resource in a lower protection class, the Reserve is set at a level which will still afford protection to the resource, but without the benefit of the buffer which is provided by caution.

To assume that a "higher" Reserve necessarily means that only a greater quantity of water is allocated to protect the resource is somewhat simplistic. The assurance, or reliability, of water, especially under extreme climatic conditions, is just as critical an aspect of the Reserve as the quantity and quality.

### 1.3.6 Protection-based classification of water resources

Depending on the level of risk which is acceptable for the resource, Resource Quality Objectives could be set at any level on a continuum between no protection at all (certain damage) and maximum protection (no risk). In practice, the decision on the level of protection which is desirable, the subsequent setting of objectives to reflect that decision, and the control and regulation of activities impacting on a water resource, can be streamlined and facilitated by the implementation of a protection-based classification system.

Under a **national protection-based classification system,** water resources can be grouped into classes representing different levels of protection. The risk that can be accepted in each class is related to the level of protection required for that class. This provides a nationally consistent basis and context for deciding on an acceptable level of short term risk, against the requirements for long term protection of a water resource. For water resources which are especially important, sensitive, or of high value, little or no risk will be acceptable, and they will be assigned a high protection class. In other cases, the need for short to medium term utilisation of a water resource may be more pressing: the resource will still be protected, but be assigned a class which reflects a higher risk. In addition, certain activities or impacts would be regulated or controlled to a certain degree in each class. Some impacts might be prohibited entirely in the highest protection class.

A protection-based classification system comprising a few protection classes is recommended. The highest class, requiring the greatest level of protection, and allowing no risk to the Resource Base, includes "special" water resources of very high value. These resources might be special in terms of their conservation importance, or because they support very important and sensitive uses. In some countries, the term "heritage rivers" is used to denote the highest class. The lower classes reflect slight risk, moderate risk or high risk of damage to the Resource Base, and the Resource Quality Objectives in each case reflect the level of risk associated with the class.

Overall, the assignment of a specific class to a water resource gives a clear message to both users and impacters regarding the social, economic and ecological value of that water resource. The classification represents a vision of how people feel their water resource should be managed, in terms of resource quality.

### 1.3.7 The process of classifying a resource and setting objectives

Classification of a water resource, and the subsequent derivation and setting of objectives, should ideally be undertaken in a formal process of negotiation and consensus-seeking among all stakeholders. Stakeholder groups which should be represented in this process include water user sectors, industrial sectors, agricultural sectors, public sectors, special interest groups, local and regional government, as well as other government departments responsible for resource development and for resource protection. Ideally, the process of classification and setting of objectives should be conducted within an integrated catchment management framework. The representation of local interests in this process is very important. However, since water is managed as a national resource, there should also be representation of regional and national interests, perhaps through the formal participation of regional and national government and regulatory agencies.

Through a process of classification, all stakeholders, including water users, impacters and the regulatory agency, can come to agreement on the level of protection which will be given to the resource, in full understanding of the implications for the degree of utilisation which can be sustained, and for the kinds of impacts which are acceptable.

### 1.3.8 Setting objectives on the basis of acceptable risk

The need for caution and the desire to prevent unintentional exceedance of the limits of sustainable utilisation is recognised as cornerstones of the policy of protection. The approach which has been adopted is that of setting limits on the basis of acceptable risk. As a result the resource protection policy requires that quantifiable Resource Quality Objectives be set which reflect our understanding and acceptance of **a particular level of risk** of exceeding and possibly causing irreversible damage to a water resource.

The extent of the accepted risk is related to the value or importance that is placed on a specific water resource. Some resources will have a very high value either because of their ecological importance and value, or because of their sensitivity to certain forms of utilisation, or because of a need to maintain long term reliability of the services provided. In this case, especially stringent Resource Quality Objectives would be set to minimise the risk of irreversible damage. For other water resources, short term needs for water or economic development might be so important or so urgent that a greater risk of exceeding the "point of no return" would be accepted with greater risk of failure of the resource, in exchange for allowing more utilisation. However, the particular level of risk should be accepted by all stakeholders, including impacters and water users, with a clear and common understanding of the possible long term consequences.

Adopting a risk-based approach 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. For example, a concentration of a toxic chemical which poses only a slight risk to a particular ecosystem in one geographical region may result in a much higher risk in another geographical region, depending on the resilience of the adapted ecosystem, the background quality of the water, and the natural flow regime. It is important to recognise that it is the concepts of levels of risk, and levels of protection, which are nationally applicable, rather than the objectives themselves. Only in a few instances, e.g. such as for persistent toxic substances, is it practical to set numerical objectives applicable to all water resources of a particular class anywhere in the country.

## 1.4 THE SCIENTIFIC BASIS FOR RESOURCE QUALITY OBJECTIVES

### 1.4.1 Objectives for ecosystems and water users

Resource Quality Objectives are scientifically derived criteria, based on the best available scientific knowledge and understanding. They represent the best assessment of the resource quality necessary to provide a desired level of protection for a water resource, with a particular degree of assurance or risk. Resource Quality Objectives will be derived for individual water resources, such as river reaches, sub-catchments, estuaries, coastal marine waters, wetlands and groundwater resources according to DWAF policy statements, methodologies and publications.

For aquatic ecosystems, Resource Quality Objectives can be derived from measurements and an understanding of ecosystems in field and laboratory conditions, especially ecosystem responses under stress induced by changes in water quantity, water quality or habitat integrity. For recognised water users, the primary focus of objectives are on water quality aspects. Objectives for water quality for these water users are based either on a scientific understanding of the direct physiological effects of changing water quality (e.g. effects on human health, damage to a sensitive crop, toxicity to livestock etc.), or on assessment of economic impacts (e.g. the cost of increased water treatment, or the loss of productivity).

### 1.4.2 Framework for setting Resource Quality Objectives for aquatic ecosystems

Resource Quality Objectives for aquatic ecosystems are based on four key aspects: water quantity, water quality, habitat integrity and biotic integrity. Table 1.1 shows a framework within which numerical objectives for each of these aspects can be derived. Each aspect is discussed in more detail sections 1.4.2.1 to 1.4.2.5.

**Table 1.1** Framework for setting numerical Resource Quality Objectives on the basis of a classification system

| Class | Water quantity | Water quality | Aquatic habitat | Riparian habitat | Biota |
|---|---|---|---|---|---|
| A | Natural variability and disturbance regime: Allow negligible modification. | Negligibly modified. Allow negligible risk to sensitive species. Within Aquatic Ecosystems TWQR for all constituents. | Allow negligible modification from natural conditions. Depends on the in-stream flow and quality objectives which are set. | Allow negligible modification from natural conditions. Control of land uses in the riparian zone in order to ensure no modification (e.g. no disturbance of vegetation within set distance from banks) | Negligible modification from reference conditions should be observed (based on the use of a score or index such as SASS). |
| B | Set in-stream flow requirements to allow only slight risk to especially intolerant biota. | Use Aquatic Ecosystems TWQR and CEV to set objectives which allow only slight risk to intolerant biota. | Allow slight modification from natural conditions. Depends on the in-stream flow and quality objectives which are set. | Allow slight modification from natural conditions. | May be slightly modified from reference conditions. Especially intolerant biota may be reduced in numbers or extent of distribution. |
| C | Set in-stream flow requirements to allow only moderate risk to intolerant biota. | Use Aquatic Ecosystems TWQR, CEV and AEV to set objectives which allow only moderate risk to intolerant biota. | Allow moderate modification from natural conditions. Depends on the in-stream flow and quality objectives which are set. | Allow moderate modification from natural conditions. | May be moderately modified from reference conditions. Especially intolerant biota may be absent from some locations. |
| D | Set in-stream flow requirements which may result in a high risk of loss of intolerant biota. | Use Aquatic Ecosystems TWQR, CEV and AEV to set objectives which may result in high risk to intolerant biota. | Allow a high degree of modification from natural conditions. Depends on the in-stream flow and quality objectives which are set. | Allow a high degree of modification from natural conditions. | May be highly modified from reference conditions. Intolerant biota unlikely to be present. |

*1.4.2.1 Water quantity objectives*

Numerical objectives for water quantity will be set on the basis of levels of risk outlined in Table 1.1. For example, in rivers a daily flow regime can be determined to ensure the provision of in-stream habitat which offers only a slight risk to especially intolerant species (class B). If a greater level of risk is acceptable, e.g. a class C, then the flow regime might reserve less flow overall, or less flow at critical periods, or some combination of daily flows that reflects a level of moderate risk to intolerant biota.

The water quantity objectives influence issuing of licenses for discharge. For example, assuming it was decided that class B streams should not be made perennial if their natural flow regime follow a seasonal pattern. A waste discharge into a seasonal class B stream should not be of such a volume that stream-flow in normal dry or low flow periods will be significantly modified. License conditions might state that the waste volume could be, at most, a certain percentage of the natural low flow in the stream during the dry season.

*1.4.2.2 Water quality objectives*

The water quality criteria for aquatic ecosystems (DWAF, 1996) have been derived according to the concept of acceptable risk. For toxic substances, the Chronic Exposure Value (CEV) represents the concentration at, or above, which there is a significant risk of chronic toxicity to 5% or more of aquatic organisms. The Acute Exposure Value (AEV) represents the concentration at, or above, which there is a significant risk of acute toxicity to 5% or more of aquatic organisms. The No Effect Range (NER) represents the concentration at which no adverse or toxic effects should be expected in aquatic biota.

Numerical water quality objectives for each water resource class can be set to maintain a desired level of risk, from no risk in class A, to high risk in class D. Waste license conditions, or non-point source controls, should be set to achieve these objectives.

There has been considerable development of the conceptual basis for ecological risk analysis and assessment, both internationally and locally. These concepts will be used in the derivation of numerical water quality objectives within the framework of a protection-based classification system.

*1.4.2.3 Habitat integrity objectives*

Numerical or narrative objectives for in-stream habitat can also be set according to acceptable levels of risk to ecological integrity. The extent, distribution, type and integrity of in-stream habitat is strongly dependent on the water quantity and water quality objectives which are set. However, objectives must be derived for other factors which influence in-stream habitat. For example, where excessive soil erosion in the catchment increase in-stream sedimentation rates to an unacceptable level, regulation of the impacts of land use practices may also be an aspect of the Resource Quality Objectives. Another example of in-stream habitat modification is through the impacts of sand winning, and objectives will be set to regulate these impacts and subsequent rehabilitation.

Riparian habitat is more at risk from land use practices, such as construction, river diversion, ploughing on river banks and urban development. Numerical or narrative objectives will be set to ensure the appropriate extent, distribution, type and integrity of riparian habitat, to maintain an acceptable level of risk to biota which rely on the habitat.

*1.4.2.4 Biotic integrity objectives*

Measures of biotic integrity are usually integrated indicators, such as the SASS (invertebrate) and fish indices. Biotic integrity is almost entirely dependent on the achievement of the appropriate objectives for water quantity, water quality and habitat integrity. Objectives in terms of scores can be set for biotic integrity, on the basis shown in Table 1.1, but the achievement of these objectives can only be assured through maintenance of an appropriate abiotic template (water quantity, water quality and habitat integrity).

*1.4.2.5 Integrated Resource Quality Objectives*

Although objectives have been set for flow, quality, habitat and, more recently, the biotic aspects, usually the objectives for flow and habitat are set independently of the water quality objectives. Taking the integrated water environment as a basis for resource management, it will now be necessary to derive integrated objectives for the entire water resource.

For example, if a river has sufficient ecological and use-related importance to be assigned to class B, then the in-stream flow regime for that resource should be set such that there is only a slight risk of stress or loss of especially intolerant biota (see Table 1.1). However, if biota are already slightly at risk due to modification of the natural flow regime, this should be taken into account when setting the water quality objectives. A "slight risk" due to flow modification, superimposed on a "slight risk" due to a change in background water quality, should not result in a "moderate" or "high" risk to biota, as the ecological integrity cannot be maintained at class B status.

Likewise, water flow and quality objectives set for biota will also need to take into account the flow and quality requirements of the in-stream and riparian habitat. Furthermore the added risk to biota if their habitat is degraded due to land use activities (such as ploughing within the riparian buffer strip, or soil erosion and subsequent deposition of sediment in the stream channel) will also need to be considered.

# CHAPTER 2
# RISK AND RELATED CONCEPTS

## 2.1 INTRODUCTION

### 2.1.1 Assumptions

There is a series of linked and very fundamental assumptions which underlie the conceptual framework for setting risk-based Resource Quality Objectives in the context of a classification system:

- Utilisation of a water resource (according to the broad definition of water use) introduces stressors to the resource in the form of changes in quantity, chemical composition and habitat and biotic integrity.
- Introduction of stressors to the resource leads to a degree of modification of the structure, function and/or diversity of an ecosystem from unimpacted (reference) conditions.
- Modification of structure, function or diversity of the pristine ecosystem causes a degree of ecological stress which can be estimated directly or indirectly with available tools and assessment procedures.
- A functional relationship exists between the degree of stress and the response of the ecosystem components. More specifically, that the level of effect can be mapped by the level of stress. This mapping will reflect the characteristics of the ecosystem (see below). In the context of risk it is necessary to determine an end-point that has both ecological meaning and can be utilised by management (see later).
- It is possible to define a level of protection that is acceptable to scientist and user alike.
- Numerical site-specific Resource Quality Objectives derived represent the conditions to be maintained in the water resource in order to provide the desired level of protection.
- Resource Quality Objectives provide management goals from which an acceptable degree of modification from reference conditions can be set and which can determine the limits of impacts of water utilisation to be allowed.

### 2.1.2 Why use risk?

The basic human needs and the needs of aquatic environment may, under certain circumstances, have conflicting requirements. Water resource management will need to find an optimal balance between resource use and/or discharge requirements and the protection of sustainability of the resource. This optimisation process is further complicated by the disparate measures or units in which requirements are specified for the users and the ecosystem. There is, therefore, a need to facilitate this process by finding a common basis for expressing requirements.

### 2.1.3 Factors impacting water resource Ecological Reserve status assessment

The Ecological Reserve is determined by the dynamics and kinetics of interactions of aquatic animals, plants and processes which determine the function, composition and diversity that characterise the ecosystem. Water quality management objectives, associated criteria and assessment approaches for a resource must reflect the inherent ecosystem characteristics if they are to achieve their goal.

Some fundamental characteristics of the ecosystems which need to be considered include:

- A variety of stressors may be at work at various spatial and temporal scales and yet result in the same unacceptable effect. For example, a fish species may disappear from a river either because of severe chemical contamination, over-harvesting of the species, impairment of crucial breeding habitat or simply because there is no water in the river. The event "disappearance of an expected fish species at point A" may be due to any combination of the above factors. The absence of water in the river may be due to stress at a scale larger than the scale of observation such as a global weather pattern or a more local abstraction upstream of point A. Contamination, on the other hand, may originate from a highly localised point source.
- There is an innate and practically irreducible inter- and intra-specific variability in biotic response to a given stressor. Biotic systems are characterised by variability (O'Niell *et al.*, 1980; Kooijman, 1987; Brown, 1993). There is an intra-specific variability in individual response to a stressor within a population as well as an inter-specific stochasticity in response at a community or ecosystem level.
- There are limits to the scientific certainties about any given natural biotic system which impact, *inter alia,* on the certainty of cause-effect relationships in a particular system. Uncertainty is largely a characteristic of the observer and their deductive processes. Since modelling, whether conceptual or mathematical, often forms a part of the deductive process, uncertainty may derive from a) uncertainty in future input to the model, b) uncertainty in structure and parameters of the model and c) uncertainty in the application and validity of the model which may well be reducible on presentation of more, or better, information. The impact of model uncertainty is so severe that the use of quantitative (usually deterministic) predictive models is disparaged by some biologists (e.g. Fryer, 1987). According to Holling (1996), there is "an inherent unknowability, as well as unpredictability, concerning the ecosystems and the societies with which they are linked"(p1).
- In many natural ecosystems there is a dearth of detailed data about structure, function and composition (e.g. Cairns, 1986: Landres *et al.*, 1988; Munkittrick and McCarty, 1995). Ecological knowledge is often descriptive rather than quantitative.
- Response of organisms to stressors is normally continuous and discontinuities are normally an artefact of the scale of observation. The variability observed in the response of organisms is generally accepted to be derived from a distribution of some surrogate of susceptibility. If the test population is large enough or the method observation discerning enough, the response of the population is essentially continuous (e.g. Hewlett and Placket, 1952; Hathway, 1984 ).

The second, third and fourth points suggest that a deterministic approach, in which only one outcome is expected when stress is applied, is impractical. For a given level of stress applied to an ecosystem, the effect expected for a given individual is impossible to predict uniquely due to the variability of individual response. The response variability alone would cause the expectation of effect to be stochastic. Superimposed on individual variability within a population is the limited knowledge of detailed structure and function of the ecosystem. There are a number of theoretical approaches by which such uncertainty can be handled, including interval analysis, probability theory and possibility theory (Klir and Yuan, 1995).

The characteristics described argue that the use of a probabilistic tool, such as risk, in resource management is appropriate. Aspects that need particular attention and are discussed in detail are listed:

- The **assessment paradigms** that are used in environmental assessment and which will determine the nature of the risk tools.
- The ***concept of risk***, as a tool in water resource management due to the characteristics of the aquatic ecosystems, its **definition** and the **interpretation of probability** as a crucial aspect in the interpretation of risk.
- The conceptual **management framework** that utilises risk technology to its full advantage.
- It will be shown that the concept of risk can be interpreted and used in several ways that is at least partially, if not entirely, dependent on **policy decisions**. Some suggestions will be put forward for consideration.
- The ability to assess the attainment of the Resource Quality Objectives is dependent on the ability to perform **ecological risk assessment**. Although risk can be assessed retrospectively, it is likely that water resource management will also have a need to apply the objectives in a predictive mode (e.g. the issuing of discharge or abstraction licences). A brief overview of some of the important features of ecological risk assessment is given to demonstrate the interconnectedness of the objectives and the risk assessment process, with particular reference to the selection of measurement end-points.
- One of the key features of a risk assessment, which often carries the biggest contribution to overall uncertainty, is the **stressor-response relationships** which are used to estimate the probability of effect. A review of the state of stressor-response relationships was discussed at a workshop to determine the viability of establishing this vital link in the risk assessment process (see Part 2: Specialist workshop 1).

## 2.2 ASSESSMENT PARADIGMS

The paradigm in which an assessment is approached may result in the expectation of the acceptability or unacceptability decision as a function of the decision process displaying either discrete quantal steps (analogous to the quantum phenomena in atomic physics) or a continuum (Figure 2.1). The quantal assessment paradigm (QAP) and the continuous assessment paradigm (CAP) were described by Suter (1990) as the "hazard assessment" and "risk assessment" paradigms respectively. The characteristics that distinguish these two alternative paradigms are listed in Table 2.1. In practice neither paradigm is used in its pure form in environmental and ecological assessment, with suitable characteristics often being selected according to the situation specific needs.

Both QAP and CAP assume that the environmental safety of a substance is based on the relationship between the response of an organism/population/community/ecosystem and the extent of exposure. This differs in principle from assessments based on some quality of the chemical (e.g. carcinogenicity), on analytical chemistry (e.g. "should not be detected") or on technology (e.g. "best available technology required"). None of these alternatives actually considers whether environmental effects are present or what the magnitude is or should be.

**Table 2.1** Characteristics of environmental hazard assessments and risk assessments (adapted from Suter, 1990)

| *Characteristic* | *Hazard Assessment* | *Risk Assessment* |
|---|---|---|
| Type of result | Deterministic | Probabilistic |
| Scale of result | Dichotomous | Continuous |
| Regulatory basis | Scientific judgement | Risk management |
| Risk/benefit/cost balancing | Very difficult | Possible |
| Assessment endpoints | Not explicit | Explicit |
| Expression of contamination | Concentration | Exposure |
| Tiered assessment | Necessary | Unnecessary |
| Type of models used | Deterministic | Stochastic |


**Figure 2.1** A representation of the outcome of an assessment as the assessment progresses. In the progress of the assessment, the confidence in the data increases. In this example both assessments start out with the assumption of unacceptable for a situation that is essentially acceptable.

### 2.2.1 The Quantal Assessment Paradigm

The QAP is analogous to the judicial model of pronouncing a person guilty or not guilty. It has the following characteristics:

- The basis of environmental quantal assessment is the "expert opinion" of what should constitute acceptable or unacceptable. The expert opinion may be encapsulated in a criterion value (CV). For example, assessing an environmental impact of a toxic release entails comparing the expected (or measured) environmental concentration (EC) to the CV. If EC > CV, then the hazard is unacceptable and if EC < CV then the situation is acceptable.
- A fundamental assumption of the QAP is that, given enough time and effort, the situation where the EC, for example, cannot confidently fit into either category, can be resolved (*i.e.* it can be assigned a unique outcome). In a situation where no clear unequivocal answer is available in assessing the status of an observation relative to the criterion, the hazard paradigm demands an iterative data gathering (testing and measurement) procedure until a definitive answer can be given. As more iterations are added to the process the confidence in the distinction between acceptability and unacceptability grows. Confidence here does not necessarily refer to statistical confidence, but more so to institutional or personal confidence (Suter, 1990).
- Essential to the QAP is a criterion (Figure 2.2). The assessment therefore needs to be repeated for every variable that may impact on the system being assessed.
- In the quantal assessment process there is not necessarily an explicit decision *ab initio* as to which end-points are being addressed; it does not intend to identify what is specifically expected to occur (Bartell, *et al.*, 1992) since these are implicit in the criteria. Both the process by which the expert selects the end-point (*i.e.* what might be expected to occur) and the extent to which this is possible is subjective to a degree even though it may be internally coherent. This aspect of the QAP makes the process inherently less transparent.

### 2.2.2 The Continuous Assessment Paradigm

In contrast, the Continuous Assessment Paradigm (CAP) is characterised by:

- Acceptance, *a priori*, that some uncertainties are practically irreducible and that a definite decision on yielding an acceptable/unacceptable outcome may be logically impossible. Consequently, there are decisions that may never (within the time frame of the decision-making process) have a deterministic answer and the process therefore relies more heavily on probabilistic expression.
- Accepting a continuum "grey scale" in assessment outcome. This results from its use of probabilistic assessment methods to accommodate uncertainty explicitly. It assumes that, unless there are specific mechanistic reasons, there are no discontinuities or "sharp steps" in the assessment outcome.
- Because of its probabilistic expression, the object and end-point appear explicitly in the assessment (the probability of what happening to whom).
- In most environmental assessment situations, the risk paradigm would appear to be the more objective means of decision-making. It must however be accepted that some form of human judgement can never be completely removed from the risk paradigm. For example, what constitutes a large or a small risk is often a matter of subjective judgement or policy.

### 2.2.3 Paradigms in water resource management practise

In many countries a common form of risk assessment involves calculation of the ratio that the ambient (or predicted) environmental concentration (PEC) will exceed the (predicted) no-observed-effect concentration (PNEC). It is normally assumed that when this ratio exceeds 1, that the effect is unacceptable. The situation is illustrated in Figure 2.2 where in reality PEC < PNEC but where the null hypothesis is that the PEC > PNEC. As the information on the actual value of PEC increases, the confidence of the assessor grows that the null hypothesis can be rejected. But until a less arbitrary confidence level, determined either by convention or policy or both, is reached, rejecting the null hypothesis is impossible and the situation is assessed as "unacceptable".

For this illustration it is assumed that at least PNEC is known with certainty. In reality both PEC and PNEC are subject to uncertainty and variability (discussed later) and therefore the assessment outcome is determined by a combination of:

- the uncertainty characteristics of PNEC and PEC, which may possibly, but not necessarily, be expressed statistically and which reflect confidence in the scientific data-base; and
- the test criterion, which may be expressed statistically (e.g. a specified confidence level in an ANOVA hypothesis test) but reflects the personal or institutional confidence in the evidence (Suter, 1993).

It should be clear from Figure 2.1 that as the decision criteria on the acceptability axis become more finely graded (*i.e.* there are a larger number effects used to define the effect in the PNEC in Figure 2.2), the QAP outcome more closely approximates the CAP outcome. This approach can be seen in the environmental criteria of a number of countries (Erickson and Stephan, 1985; Roux, *et al.*, 1995).

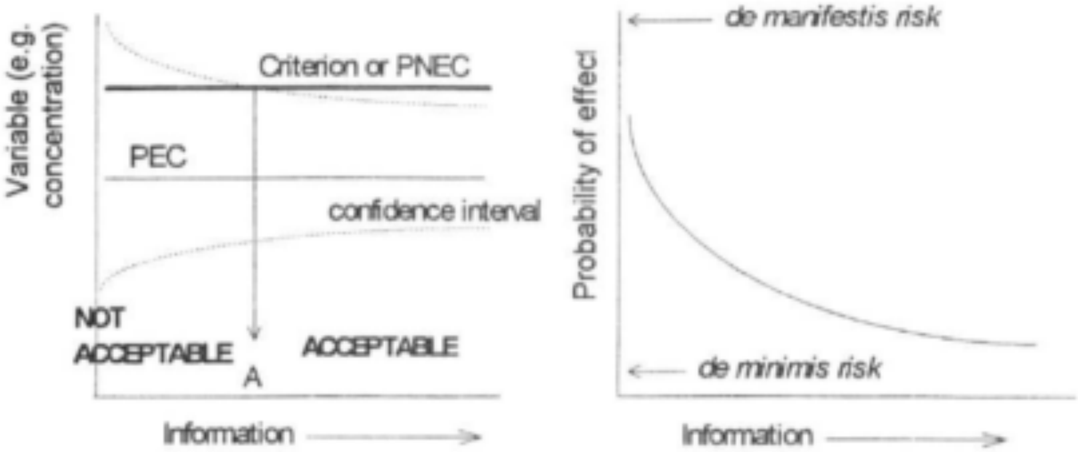



**Figure 2.2** The outcome of the assessment as a function of increase in information as the assessment progresses. The quantal assessment stops at point A when it becomes apparent that the PEC can statistically be resolved from the criterion or the predicted no-effect concentration (PNEC). In the continuous assessment, the stop criterion is determined by the purpose of the assessment. The *de manifestis* (probability $\rightarrow 1$: clearly unacceptable) and *de minimis* (probability $\rightarrow 0$: clearly trivial) risks are the only inherent cut-off points in this kind of assessment.

In law enforcement the QAP is the preferred paradigm since its output is amenable to legal and judicial requirements. However, in the earlier phases of management where various stressor sources might need to be considered, the CAP may be preferred. It is unreasonable to assume that a situation is completely acceptable up to a point beyond which it becomes completely unacceptable. There is a need to establish a process by which the ecological response criteria can be derived in keeping with the non-deterministic nature of the underlying ecological processes.

In the South African Water Quality Guidelines for the protection of Aquatic Ecosystems (DWAF, 1996), three levels of criteria are defined. The Acute Exposure Value (AEV) nominally represents a level of stressor at which 50% or more of the population, or up to 5% of aquatic species, could be killed on exposure for a few days. The Chronic Exposure Value (CEV) nominally represents a level of stressor at which up to 5% of aquatic species may experience some form of sub lethal impact over a period of some weeks. The Target Water Quality Range (TWQR) represents a level where no effects are expected. The shortcomings of these criteria in terms of the Ecological Reserve status assessment relates to the disparate time scales within which they are applicable and the disparate effect scales.

In the National Water Act (Act 36 of 1998), specific provision is made for classification of water resources. This implies that there is an explicit need for a quantal assessment of the resource. *It is important to see that the use of classification in the context of the Ecological Reserve is an administrative management tool designed for specific management needs and that it does not imply discontinuity in the ecological function or stress-response domains.* Where water resource management needs may require it, the CAP may be invoked to solve a particular problem. In this process a continuum is generated which may be discretised again as needed.

## 2.3 THE CONCEPT OF RISK

The concept "risk" used colloquially has the semantic implication that "something unacceptable may happen". The colloquial use emphasises the level of hazard attached to a subject (e.g. "Mercury poses a risk to human health" or "Rock climbing is a risky sport") which is not always easy to quantify. This usage often emphasises the magnitude of the consequences as perceived within the reference frame of the observer. In this sense, an expression such as "the risk of an asteroid hitting the earth is much greater than that of being involved in a motor car accident" may be perfectly true from the point of view of consequence although the likelihood of the former happening is several orders of magnitude less than the latter. In addition, this may only be true for an observer concerned with global survival, while it may be completely false for an insurance broker. The problem in using the term risk in this way, is that: a) it is difficult to find a common basis for comparing such diverse consequences as the impact of an asteroid and a motor car accident, and b) there is a strong element of subjectivity in the assessment.

In this document a more technical usage is preferred. In the more technical usage, "mercury", "rock climbing", "asteroid impact" and "motor car accident" would be referred to as "hazards" rather than "risk". There is also an extensive discipline dealing with a more technical usage of "risk".

### 2.3.1 Definition of risk

The concept of **risk** has been widely used in various disciplines such as engineering, epidemiology, sociology, toxicology and economics. The technical concept of risk was defined in 1901 for the actuarial sciences as "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1990, p) or the probability of observing a specified (unacceptable) effect as a result of a toxic chemical exposure (Bartell, *et al.*, 1992) or simply the probability of experiencing an effect of a hazard (Haas, 1988). In essence, whether explicitly or implicitly, the technical usage of the term risk contains elements of:

- the existence of a hazard or stressor,
- probability (as one of a number of means of quantifying variability),
- a target or object and
- an undesired effect.

**Risk assessment** is an array of techniques that is primarily concerned with the estimation of the probabilities and magnitudes of events. The probability element implies that in principle there is a continuum of risk from infinitely small (practically zero) to infinitely high (practically 100%). Due to practical limitations, coarser resolution (e.g. small, moderate, or high) is also used.

### 2.3.2 Interpretation of probability

Interpreting the term "risk" and "probability" has a fundamental impact on the approach to, and application of, risk methodology. The more traditional view of probability, deriving from the work of the 17th to the 19th centuries, emphasises probability as the limiting frequency of a large number of observations; this is also referred to as the "frequentist" approach (Jaynes, 1996). This aspect of probability emphasises the ontological nature of probability: probability as a real, mathematically calculable entity expressing a characteristic of data.

However, even this limiting value of data may only be a description of a state of knowledge about the data relating to some measure of confidence and therefore epistemic in nature. Consequently, De Finetti (1990) describes probability as a subjective expression needed to project from the domain of uncertainty, by the means of prevision, to the domain of certainty. "Prevision... consists in considering, after careful reflection, all the possible alternatives, in order to distribute among them, in the way which will appear most appropriate, one's own expectations, one's own sensations of probability" (De Finetti, 1990).

Probability, and by association risk, can be seen as a specific combination of situation and observer. The implication of this is that risk is explicitly bound to a given situation. It may be argued that an absolute risk does not exist and that it needs to be placed in a frame of reference to have any meaning.

Regulatory decision making in the field of ecology is largely dependent on a descriptive conceptual knowledge of ecosystems, often only supported by patchy observation. The use of frequentist statistics is largely ruled out by a lack of specific data. The expert prevision pertaining to a specific situation needs to be considered. Ecological risk, as a probabilistic expression, is essentially a subjective estimate of the likelihood of an effect, a prevision based on the assessor's best available knowledge of, and expertise in dealing with, unobserved events in a complex system.

### 2.3.3 Variability, uncertainty and vagueness

*2.3.3.1 Variability*

Biotic systems are characterised by variability (O'Niell *et al.*, 1980; Brown, 1993). There is an intra-specific variability in individual response to a stressor within a population as well as a variable inter-specific response at a community or ecosystem level. It is useful to distinguish between uncertainty and variability since their impact on the confidence of an assessment may be very similar while the sources may be very different (Frey, 1993). Variability is an **irreducible characteristic of the observed system**. It does not decrease when more data becomes available but it is better characterised.

*2.3.3.2 Uncertainty*

Uncertainty is largely a **characteristic of the observer and his/her deductive process**. Since modelling, whether conceptual or mathematical, often forms a part of the deductive process, uncertainty may derive from a) uncertainty in future input to the model, b) uncertainty in structure and parameters and c) uncertainty in the application and validity range of the model and may well be reducible on presentation of more or better information. Both uncertainty and variability have the effect of introducing a degree of fuzziness in a predictive mode. The impact is so severe that the use of quantitative (usually deterministic) predictive models is disparaged by some biologists (e.g. Fryer, 1987). The dilemma for water quality management in this context is the legal mandate which has been entrusted to it for the protection of the Ecological Reserve requires both predictive and analytical capability in an apparently chaotic domain.

Given the observed uncertainty and variability, there is no deterministic outcome to ecosystem assessment. In dealing with ecosystem assessments it is necessary to apply techniques that explicitly recognise the stochastic elements of the system. The definition of the Ecological Reserve should therefore explicitly recognise the impact that ecosystem characteristics have on the ability to predict and analyse the effects in that ecosystem.

### 2.3.4 The compatibility of risk with the Reserve concepts

The Ecological Reserve can be conceptualised as an amount of "buffering capacity" to the Resource Base. The Resource Base is the minimum critical state of the ecosystem necessary to sustain its continued functioning. The variables that describe this critical state may include water quantity, physical and chemical composition of the water, integrity of the biota and habitat, availability of refugia etc. These variables are naturally temporally and spatially variable and their interactions are often both variable and uncertain. Furthermore, there are not necessarily any unique values of the variables that describe the reference or "pristine" state, but rather a set of variables that is in a dynamic internal (intra-set) equilibrium. Likewise, the Resource Base is a dynamic combination of variables. Although the individual variables are dynamic, the system state is stable. As the variables deviate further from their "pristine" values, there is an increasing expectation that the system state will change. The magnitude of this expectation is determined by the assessor's state of knowledge, experience, viewpoint etc. and is situation specific. It is therefore reasonable to represent this expectation as a probabilistic risk of change of ecosystem state.

Given continuity, uncertainty and variability observed in ecosystems, the implications for the Ecological Reserve are:

- The Resource Base is essentially an uncertain, and possibly vaguely defined set of stochastically dominated conditions which is best described in terms of an expression of likelihood (e.g. probabilistic or possibilistic) terms.
- A binary (or ternary or any other n-nary) assessment result can only be attained if a set of assumptions or values are used to discretise the assessment domain.

These have significant implications for resource management. It determines how risk is formulated, what the minimum database should comprise of for management decisions and how dependent the management process is on social, economic and policy inputs. In order to clarify these issues, it is necessary to consider the following:

- distinguishing between vagueness and stochasticity
- handling variability and uncertainty
- preliminary considerations in setting risk objectives.

*2.3.4.1 Distinguishing between vagueness and stochasticity*

As shown above, it is true that no unequivocal, deterministic pronouncement regarding ecosystem processes can be made. The Resource Base is therefore also an entity about which no deterministic pronouncement is possible.

A crucial question in formulating Ecological Reserve objectives is whether the Resource Base is vague or stochastic. In other words is it an ill-defined deterministic quantity, or is it a precise quantity random in value? The same characteristic will be reflected in the Ecological Reserve and this needs to be recognised in setting the risk objectives as it will impact on the way the risk objective is formulated. The reason is that the more traditional probabilistic risk (a product of probability theory, see below) is meant to deal with stochasticity but is ill equipped to deal with vagueness. The latter can better be dealt with by possibility theory, a product of fuzzy logic.

*2.3.4.2 Handling variability and uncertainty*

While both variability and uncertainty can in practise be dealt with by statistical techniques, their interpretation and management requirements are very different.

Variability has been described as a property of nature and represents the diversity and heterogeneity in a well characterised population. Uncertainty, on the other hand, represents partial ignorance or lack of information about a poorly characterised phenomenon or model and is a property of the user of the information (Burmaster, 1997). In resource management both are present and are often convoluted. An observation of a biotic system (and consequently of any determinant of the Ecological Reserve) may be a convolution of:

- response variability due to diurnal, seasonal or stage cycles;
- individual variability in susceptibility to environmental conditions;
- uncertainty due to inadequate experimental design to account for the above;
- uncertainty due to imperfect observation technique;
- uncertainty due to lack of knowledge regarding the relationship between the observation measures (metric) and the characteristic to be evaluated.

*2.3.4.3 Preliminary considerations in setting risk objectives*

Management objectives which do not take cognisance of uncertainty and variability are likely to hamper optimisation of the objectives. Theoretically, it may be impossible to manage for "no effect" as a threshold of effect may be nothing more than an observation artefact. The inability to distinguish "effect" from "no-effect" may be due to experimental design or method of observation. Objectives need to be described in terms of acceptable level of effect while the term "acceptable" needs to be defined rigorously.

In order to combine the technical concept of risk with the concept of an Ecological Reserve, it is necessary to:

- Define an *object* for the risk (*i.e.* "What is at risk?")
- Define and *end-point* for the risk (*i.e.* "A risk of what?)
- Define a risk *reference condition* (*i.e.* "The risk pertaining to the given situation is more or less than the reference situation")
- Identify the *hazards/stressors* (*i.e.* "The risk due to A and/or B is ...")
- Identify and describe the *variability and uncertainties* in the stressor-ecosystem interactions
- Describe the *state of knowledge* about the ecosystem and its expected interactions with the stressors

The above are steps in the problem formulation phase of a generic risk assessment procedure as described by USEPA (USEPA, 1997) and Suter (1993). While risk assessment has as its goal estimating risk, setting or derivation of risk objectives have these steps in common with risk assessment. The detail of generic ecological risk assessments are considered in the next section.

## 2.4 ECOLOGICAL RISK ASSESSMENT

### 2.4.1 Individual stressor risk

"Risk Assessment" refers to a range of techniques used to estimate the probability of an effect of a stressor and has been formally defined as (Suter, 1993): "the process of assigning magnitudes and probabilities to the adverse effects of human activities or natural catastrophes". Ecological Risk Assessment (ERA) "evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors" (USEPA, 1992a).

ERA specifically deals with the probability of stress due to individual stressors. Well-established procedures exist to calculate these risks and a fairly generic procedure is described in Appendix 2. Key features of this procedure are:

Step 1: Defining the problem. This includes selecting the target at risk, the assessment and measurement end-points. Part of this process involves translating the human value issues into a scientific ecological problem.

Step 2: Assessing the likelihood of effect of the stressor on the target biota. This facet of the assessment concerns itself largely with estimating the probability of effect from suitable stressor-response relationships. Extensive literature and databases exist for a variety of toxic stressors and for certain end-points (e.g. the AQUIRE, ASTER and ECOTOX databases maintained at the USEPA's Laboratories in Duluth, Minnesota).

Step 3: Estimating the probability that the target biota will be exposed to the stressor. For chemical stressors this is an exercise in transport and fate modelling for which numerous predictive models exist (USEPA, 1996). The output of this process is either a probability of a specific level of exposure or an exposure probability distribution.

Step 4: Risk integration and characterisation. In this phase the output of Steps 2 and 3 are integrated by the techniques from the domain of probability theory and statistics and the result is interpreted in terms of the problem as defined in Step 1.

In its simplest form, the result of risk assessment is a probability of a given effect (as selected in Step 1) to a given end point on selected target biota due to a given level of stressor applied to generalised habitat of the biota.

The predictive mode of risk assessment is often used in the context of stress evaluation when the impact of the stressor is still unknown or there is a change in level of stressor. Retrospective risk assessment is used when an effect has been noted, but there is uncertainty about the source of the stress (Suter, 1993). The answer provided by retrospective risk assessment is the probability or probabilities that a stressor(s) has been the cause of the observed effect.

### 2.4.2 Assessment endpoints for the Ecological Reserve

The assessment end-points are key issues in risk assessments. They should satisfy (Suter, 1993):

- societal relevance
- biological relevance
- unambiguous operational definition
- accessibility to prediction and measurement
- susceptibility to the hazardous agent

These assessment end-points are distinguished from the measurement end-points which are field operatives input to the assessment and would normally be figures such as LC50s or NOECs. The scale of assessment and measurement end-points are usually quite different. Assessment end-points refer to characteristics of populations and ecosystems while measurement end-points often refer to laboratory scale or small-scale field measurements. For example, for the derivation of water quality criteria the assessment end-point might be to protect on average 95% of the species (Stephan, *et al.*, 1985, Roux, *et al.*, 1997) or to protect 95% of the species all the time (Kooijman, 1987), or to protect 95% of the species 95% of the time (HCN, 1989). In the first case the implication is that up to 5% of the species may on average be lost from the system and this may well include highly, both societally and ecologically, valued species. The second alternative would very likely be over-protective while the third alternative may appear to be reasonable but it is based more on policy than on scientific evidence (Suter, 1993).

The choice of end-point is also an important part of the risk-based objective as it determines both the practical value and viability of the objective. It is likely that each ecosystem will have its own unique end-points which are characteristic of that system. As a default, an end-point which could be set as a matter of policy is the protection of 95% of all species, 95% of the time.

### 2.4.3 Stressor level (magnitude) profile

The response of an organism will depend on the extent to which susceptible organs, physiological processes or molecular receptors come into contact with a particular stressor. This is defined in toxicological literature as the "dose". The actual dose at the site of action is often difficult, if not impossible, to determine due to a lack of knowledge of the specific site of action. As a first approximation the "body burden" of the stressor is used as a surrogate for receptor dose. Generally the body burden is determined by the organism's physiology and specific uptake and depuration mechanics (Mancini, 1983; Sijm *et al.*, 1993). As a second approximation, it is assumed that the ambient level (concentration) or magnitude of the stressor is proportional to the body burden (e.g. Landis and Yu, 1995). In situation analyses it is the ambient level of stressors that are normally measured. In predictive assessments, the ambient level of stressors are modelled separately from the effects. A number of these stressor profile models exist (USEPA, 1996).

An issue in stressor level monitoring and prediction is the spatial and temporal extent of the model. In the case of ecosystem-level effect assessment, careful consideration of the scale needs to be given. While individual organisms may respond at the spatio-temporal scale of a few millimetres and a few hours or perhaps days, the ecosystem responds at a scale of perhaps tens or hundreds of kilometres and years or decades (even centuries). It is therefore important that a suitable scale of assessment or modelling is chosen. Given the difficulty in defining the spatio-temporal boundaries of ecosystems, this choice might also have to be made by assumption as a matter of policy. The "significant resource" classification framework may determine the spatial boundary while the temporal scale is not explicitly stated and may be affected by:

- the life expectancy of ecologically important or other species (if identified). The temporal scale should span several generations of such species;
- the frequency of biomonitoring in the resource; and
- the temporal framework for water resource measures in the area (e.g. the lifetime of abstraction or discharge licenses).

### 2.4.4 Stressor effect (response) relationships

In addition to the problem of disparity in scale of measure vs. scale of effect already mentioned, the assessment of effect is also affected by the problem of variability in susceptibility among species within an ecosystem. Conventionally the effect of a stressor is measured in a controlled experiment on a single or perhaps a few selected species in a laboratory environment for a period of time, as dictated by experimental considerations. In a risk assessment the scale issue manifests itself in:

- the uncertainty in extrapolation of effect for the same species from the laboratory environment to in-stream communities or ecosystems; and
- the extrapolation of effects from data gathered for a few species in controlled laboratory tests to a larger number of species in functioning ecosystems.

The merits of extrapolations which has at its core the "reductionist vs. holistic approach" debate, which is outside the scope of this document. But in order to have some basis for assigning a magnitude of biotic effect a critical review of the state of stressor-response relationships is needed.

For toxic xenobiotics the conventional approach has been to perform the "standard" aquatic toxicity tests (ASTM, 1993a, 1993b, 1993c). Extensive databases exist that list common end-points such as 24 to 96 hour LC50s, fecundity LOECs or NOECs or other acute or chronic toxicity data for a number of substances and species (USEPA, 1998).

### 2.4.5 An illustration

Using entirely fictitious figures to illustrate the risk assessment process and the use of risk-based objectives in the context of a water resource management situation, the following example is given. The human value issue in a particular situation may be summarised as: "This is a beautiful pristine area and we would like to keep it that way, but there is an important agricultural operation just upstream of this area. They are both important to us".
The resource manager's interpretation of these issues may be: "This is a Class A river, requiring negligible risk activity. To which levels should the in-stream concentration of fenthion be managed to maintain its present state. The fenthion concentration is currently at 3 µg/l and the abstraction is 18% of the MAR."
The risk team ecologist's formulation of this problem may be: "This reach is rich in biodiversity. There are some sensitive red-data species and sensitive riffle dwelling invertebrates. There is reason to believe that the system might already be stressed due to the abstraction of water from the river and the increasing sediment load due to the agricultural operation. The protection of almost all species in the system is essential."
Based on ecotoxicological, hydrological and limnological information combined with modelling, the risk assessor may conclude that: " The probability that 5% of the species may disappear due to the various stressors can be summarised as in the given table. Therefore, the risk due to fenthion remains at 1:5000. The risk due to water abstraction is at 1:100 while the risk due to siltation is now at 1: 1000 but will increase to 1:5 in 20 years at the current rate".
The recommendation of the risk assessment team to the resource manager may be: "If the fenthion concentration is kept to its current levels it should pose no problem to the ecosystem. At the moment the risk due to water abstraction is low. At the moment the risk due to siltation is negligible but in about ten years the risk due to siltation will equal that due to water abstraction and will then become moderate to high."

Based on the results of the ERA, the resource manager in consultation with a hydrologist may then argue: "The abstraction needs to decrease to 10% of the MAR for the risk to change from low to negligible, which means that water use needs to be curbed. But given the investment in the current usage of water and the cost of changing to less a water consumptive style of agriculture, the risk benefit:cost ratio is quite low so that this has to be phased in. The risk benefit:cost ratio for a decrease in siltation rate is very high (low cost to decrease the cultivation area to allow a 5m buffering strip next to the river) and could be implemented immediately".

While the real-life issues and problems are much more complex than this example, it illustrates an advantage to risk-based objectives in combination with risk assessment as a means to facilitate the resource management process.

## 2.5 A RISK MANAGEMENT FRAMEWORK

Risk assessment is here viewed as a tool which is best applied in, but not exclusively dedicated to, a risk management framework (Figures 2.3 and 2.4). The stimulus for using risk assessment as a fundamental component of environmental decision-making derives from the recognition that (Suter, 1993):

- the cost of eliminating *all* environmental effects of human activities are impossibly high; and
- regulatory decisions must be made based on incomplete scientific information.

In an appraisal of risk assessment and risk management in regulatory programmes, the Commission for Risk Assessment and Risk Management (CRARM, 1996) came to the conclusion "that it was time to modify the traditional approaches to assessing and reducing risks that have relied on a chemical-by-chemical, medium-by-medium, risk-by-risk strategy" (p) and to focus rather on the overall goal of risk reduction and improved health status. Risk assessment was developed because scientists were required to go beyond scientific observation to answer social questions about what was safe. Their suggested framework for risk management comprises six stages as shown in Figure 2.3.

This framework comprises six steps all involving stakeholder participation:

- Formulating the problem in broad context of human or environmental health and the interdependence of related multimedia problems
- Analysing the risks
- Defining the management options
- Making sound decisions
- Taking action to implement the decisions
- Evaluate the effectiveness of the implemented decisions in solving the defined problems.

Another view of the stakeholder-risk assessment-risk management process, emphasising the multidisciplinary input in such a participatory style of management, is presented in Figure 2.4. An essential aspect of risk assessment is the definition of end-points. If a participatory management style is followed then the formulation of risk objectives (or criteria) should make provision for the translation of societally valued end-points into assessment end-points and also supply the means to communicate these risk issues to "lay" stakeholders.

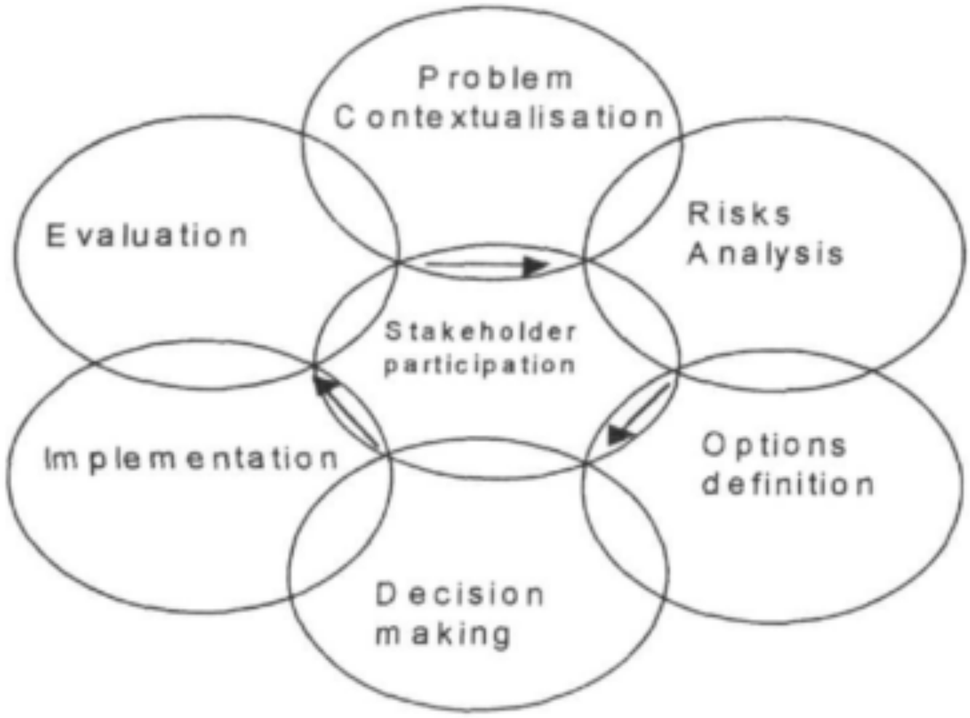



**Figure 2.3** The framework for risk management as suggested by the Commission for Risk Assessment and Risk Management (CRARM, 1996).

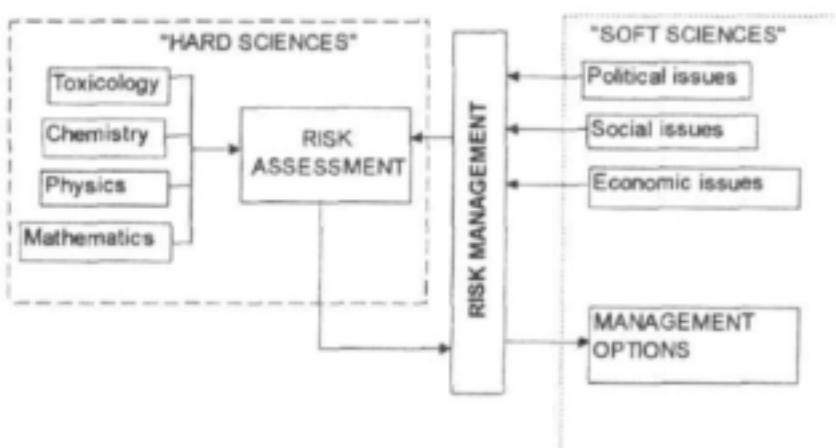



**Figure 2.4** Schematic of risk management in relation to integrating the "hard" (physical) sciences and the "soft" sciences (humanities). "Risk assessment" is here understood to refer to all risk oriented techniques available to the resource manager.

### 2.5.1 On managing hazard vs. risk

In a precautionary approach adopted by many countries where source control is viable, it is considered more advantageous to manage the hazards by removing them from the system. The hazards are ranked based on some measured or perceived scale and a more or less arbitrary decision is made which are deemed unacceptable. The basis for such decision is often the measured or expected effect, while the exposure is seldom considered explicitly. Assumptions, such as a "reasonable worst case scenario", are used to determine the management objectives of these stressors.

The advantage of hazard management described above is its relative administrative ease. Risk management requires change of style in comparison to hazard management:

- end-points are no longer implicit as in hazard management but they need to stated explicitly;
- there is a greater reliance on exposure modelling;
- there is a greater understanding of underlying science needed in risk management;
- risk management facilitates a participative management style; and
- the acceptance of non-deterministic processes in management has the advantage of greater flexibility in making management decisions.

## 2.6 ECOLOGICAL OBJECTIVES VS. MANAGEMENT OBJECTIVES

A distinction is made between ecological objectives and management objective, not because they are conflicting, but because of the difference in formulation. Ecological objectives derive from ecological goals (Figure 2.5). These goals may contain nothing more than an expression of societally or scientifically valued concepts such as aesthetic beauty or sustainable use. These goals need to be translated into quantitative or semi-quantitative objectives that can be assessed in some way such as ecological integrity or biodiversity. These ecological objectives can then be translated into the management objectives such as optimisation of discharge/abstraction, minimisation of contamination, remediation of riparian habitat. In each case objectives will have some associated metrics such as indices of biotic integrity or loads of toxics or flow requirements.

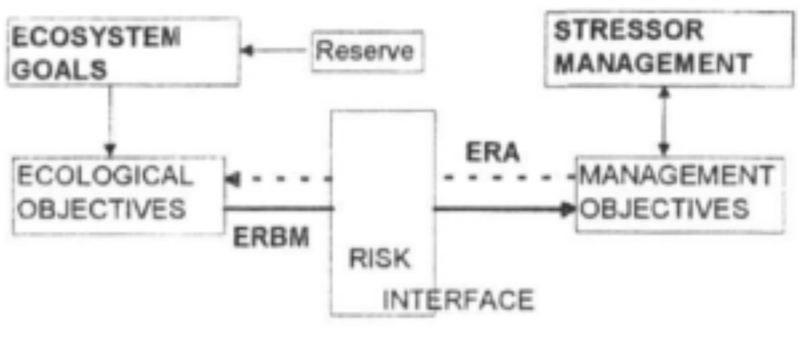



**Figure 2.5** A schematic of the ERBM process also showing the relative position of the ERA logic flow. More detail on the risk interface is shown in Figure 2.6.

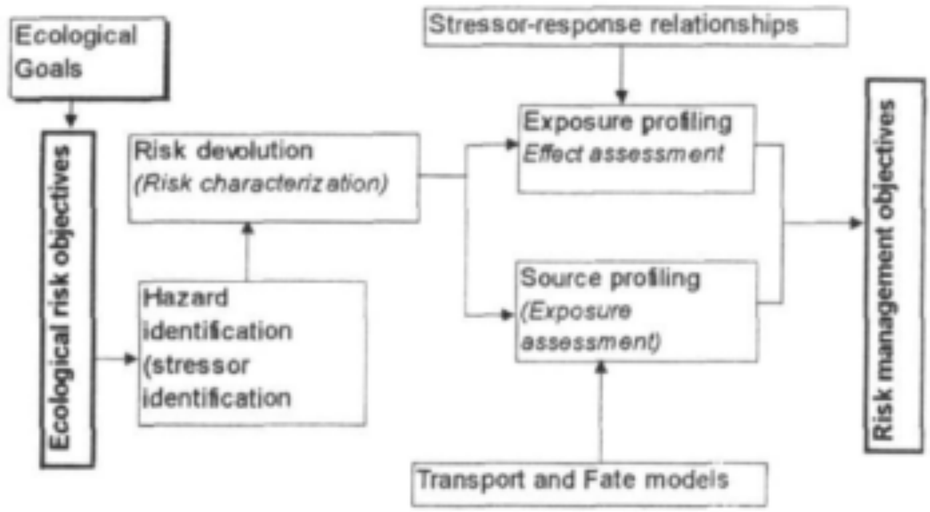



**Figure 2.6** Detail of the risk interface in the Ecological Risk-Based Management process. The phrases in italics show some of the tasks that parallel those in ERA.

Suggested detail of a risk-based management approach as means to interface ecological objectives and management objectives (Figure 2.5) is presented in Figure 2.6. Critical aspects of this model include:

- The formulation of ecological risk objectives in keeping with the goals set in the Ecological Reserve
- The means to set management goals when the stressors do not yet exist (such as when new dischargers/abstractors may impact on an aquatic ecosystem. Stressor-effect and transport and fate models may play a vital role.
- The means to derive ecological metrics reflecting risk objectives.

## 2.7 RISK OBJECTIVES VS. RISK CRITERIA

Quantal assessment provides a simple and efficient procedure for decision-making with respect to the aquatic environment even though it assumes an over-simplified model. It has the advantage of being simple to interpret with respect to law enforcement. However, in real environments its pronouncements are often scientifically indefensible. While the continuous assessment is scientifically more tenable, its regulatory application is potentially more cumbersome. A compromise is reached by creating artificial "bright lines" or benchmarks in the risk continuum and to use these in a quantal assessment mode to facilitate management decisions. The *de manifestis* and *de minimis* risk levels are commonly used bright lines. In applying this latter approach, the issue is defining the criteria for a bright line.

The assessment criterion and its interpretation are closely linked to the assessment paradigm. A criterion may have two contributory components:

- a scientifically derived component that encapsulates aspects of the observations (hopefully objective) of the system and its characteristics to which the criterion applies and which is at least semi-quantitatively measurable; and

- a component that encapsulates a value judgement on what it intends to represent or goals it intends to reach.

The degree to which these components are convoluted determines the flexibility of its use. Both components are usually dynamic and provision has to be made for adapting an assessment criterion. However, these components are affected by different underlying dynamics. The rate of change of the basic scientific component is assumed to be inversely proportional to the body of prior knowledge. The value judgements are, on the other hand, site and time specific. Value judgements used in ecological assessments may contain elements of both basic scientific knowledge but also human values relating to the ecosystem or subjective opinion. While ecologically relevant criteria must necessarily be dynamic, the flexibility (and incidentally also the transparency) of criteria can be improved by changing the point at which the most variable component is incorporated in the assessment process (Figure 2.7).

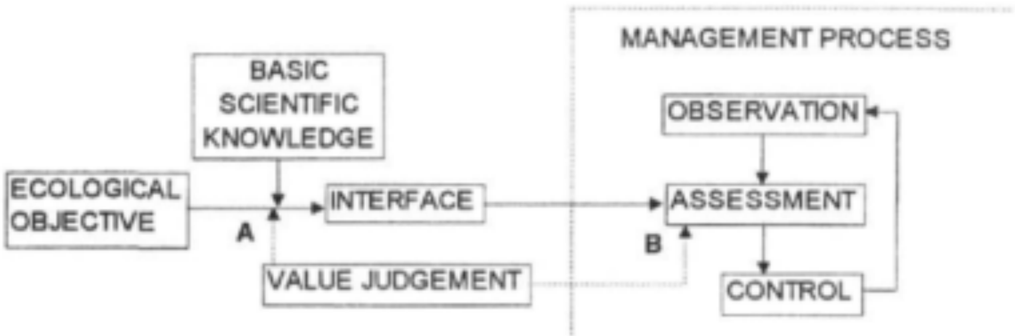



**Figure 2.7** A schematic of the options in incorporating value judgements in the resource management process for the protection of the Ecological Reserve. When the value judgement is brought into the process at A, then the interface is a conventional assessment criterion typically used in the QAP. The situation where the value judgement is brought into the process at B is described in the text.

When value judgement is incorporated into the assessment process at B (Figure 2.7) then the stress-response characteristics (variability, uncertainty etc.) should be reflected in the interface. Risk technology supplies a useful framework for expressing both the objective and the interface in a coherent format that can be readily incorporated with value judgements in the assessment step. The stressor-effect knowledge base, which relates stressor magnitude (e.g. concentration, flow rate etc.) to measurable effect (e.g. %mortality, inhibition of breeding etc.) would often be expressed in disparate units which cannot be directly compared for different stressors. If these stressor-effect relationships are translated into stressor-risk relationships then comparison and other manipulations are facilitated.

The risk management framework discussed above can be used to address the issue of assessment criteria. Through stakeholder participation value judgements can be formulated which can be used to assess management options in the light of available scientific knowledge. The concept of relative risk can be applied to compare different management options.

### 2.7.1 Defining acceptable risk

In defining acceptable risk, it must be recognised that ecosystems are subject to natural and anthropogenic stressors. This could be used in defining useful concepts in determining the acceptability of risks.

*2.7.1.1 Natural hazards*
Even under pristine conditions, ecosystems are subject to stress. This may be due to factors such as chance occurrences of catastrophic events both globally (e.g. ice ages, global warming) and locally (e.g. successive years of severe drought or catastrophic floods). Since these are outside the management domain, they pose a "baseline risk" and it should be recognised that any risk smaller than this "baseline risk" may be considered negligible. There is also a probability that a system may naturally be changing. Within the time frame of resource management, this change (hazard) may be very small although its probability may be significant. Since it normally falls outside the realm of observation it is not considered further.

*2.7.1.2 Anthropogenic hazards*
For management purposes the anthropogenic stress is of importance since it can in principle be managed. Much of the development that has taken place over the centuries has had a direct or indirect impact on aquatic ecosystems. The changes at landscape or global scale has had an impact on local aquatic ecosystems. These induced changes may have had effects, which, in retrospect, are unacceptable but practically irreversible. Other current or relatively recent human activities may also induce changes in the attributes that determine the sustainability (or resilience or integrity) of ecosystems. It is the latter type of activity that may be practically of importance to resource management. Consequently, it is the risk posed by the latter type of hazard that will generally be considered under anthropogenic hazard.

While it may be recognised that in principle ecosystems may have very few natural discontinuous response thresholds, as a matter of practical resource management, it is often required that quantal assessment of the water resource be performed (Section 2.2). As discussed above, the *de minimis* and *de manifestis* risks are conceptually natural criteria that discretise the risk continuum. Both are, however, legislative or administrative concepts determined by the *perception* of risk by the legislator or legislated community.

*2.7.1.3 De minimis risk*
The *de minimis* risk derives its name from the expression used in legal terminology: *de minimis non curat lex* (the law does not concern itself with trivials). In the risk community this term is used to refer to a level of risk that is too small to merit any further consideration. In the finality, the application of *de minimis* risk as a concept is largely a regulatory one and, therefore, its definition is largely one imposed by the legislator and may be (in purely scientific terms) arbitrarily defined. As a minimum requirement its rationale should, however, be scientifically defensible. For example, in terms of aquatic ecosystem risk, the risk posed by natural hazards may be thought of as a *de minimis* risk.

#### *2.7.1.4 De manifestis risk*

The term *de manifestis* risk has been coined to refer to a level of risk that is so large that allowing it is inconceivable. There no clear natural level of risk that could correspond to the *de manifestis* risk. It needs to be established by the legislator. Its level will probably be determined largely by perception within an anthropocentric risk reference framework.

#### *2.7.1.5 Use of bright lines*

Something along the lines of Table 2.2 might be used to define bright lines: e.g. *de minimis* risk might be defined as negligible while *de manifestis* risk may defined as high. It might be viable to obtain specific values or ranges for such categories from other studies. These descriptors need not necessarily refer to natural or environmental events but may refer also to everyday risks.

**Table 2.2** A semi-quantitative approach to risk characterisation

| *Risk descriptor* | *Qualitative description* |
|---|---|
| Negligible | Probability similar to natural global events which shape changes in the ecosystem (e.g. ice ages) |
| Low | Probability similar to natural local events which change ecosystems (e.g. severe floods, droughts) |
| Moderate | A probability of change that is clearly higher than that of natural events but which is acceptable in view of biotic uncertainties |
| High | A definite probability of change |

The important feature of criteria that discretise the risk continuum is that there is an element of perception in them. How this subjective perception is formalised into risk criteria is a matter that needs to be considered by the legislator or administrator that requires the discretisation of the risk continuum. This is an area of risk management that naturally involves community participation. An important problem in risk management relates to risk communication. Stakeholders in the use and management of water resources need to able to relate to the target risk levels. Data such as those in Table 2.3 might assist communicating and establishing risk bright lines.

**Table 2.3** Human mortality risk benchmarks for establishing and communicating risk (from Chapman and Morrison, 1994)

| *Cause* | *Probability* |
|---|---|
| Motor vehicle accident (USA) | 1:100 |
| Murder | 1:300 |
| Fire | 1:800 |
| Firearm accident | 1:2 500 |
| Electrocution | 1:5 000 |
| Asteroid/ comet impact | 1:20 000 |
| Passenger aircraft crash | 1:20 000 |
| Flood | 1:30 000 |
| Tornado | 1:60 000 |
| Venemous bite/ sting | 1:100 000 |
| Fireworks accident | 1:1 000 000 |
| Food poisoning (botulism) | 1:3 000 000 |
| Drinking water with EPA limit of trichloro-ethylene | 1:10 000 000 |

# CHAPTER 3
# ASPECTS OF THE FEASIBILITY OF APPLYING RISK-BASED OBJECTIVES

## 3.1 INTRODUCTION

The possibility of implementing the risk concepts discussed in Chapter 2, and the link to management objectives set out in Chapter 1, will determine the extent to which risk concepts can be incorporated into practical management measures within financial and manpower constraints.

The technique of Ecological Risk Assessment (ERA) draws on the disciplines of ecology, statistics, ecotoxicology, chemistry, hydrology, limnology as well as a range of other biological and physical sciences. It is unlikely that any single person will possess all the necessary skills to perform and interpret such assessments. It is often true that experts within each discipline still need some training and a measure of insight into the type and level of contributions of other disciplines if they are to become skilled risk assessors. The level of manpower in many of the crucial disciplines, such as aquatic ecology and aquatic ecotoxicology, may be inadequate to sustain the development and implementation of risk-based objectives (RBO) in resource management. An inventory and critical assessment of the skills base needed to develop, implement and audit risk-based objectives in the context of the water resource protection is needed. This has not been addressed in this study.

At a more technical level, the implementation of risk-based techniques will require some generic inputs. These relate to 1) a formulation of measurable assessment end-points, 2) assessing the effects of stressors, and 3) assessing the levels and frequencies and duration of exposure of target biota to stressors.

### 3.1.1 Formulating ecological risk objectives

The process of formulating a risk objective starts with the formulation of ecological goals. The maintenance of ecological sustainability, as pointed out in Chapter 1, may refer primarily to a management goal. This management goal has ecological implications. It would be necessary to define ecological objectives which support this management goal. An ecological goal associated with the goal of sustainability may be (for example) the maintenance of ecological integrity. This may be translated into ecological objectives such as the maintenance of diversity, structural and functional integrity.

The ecological objectives need to be translated into risk objectives. This will require:

- defining suitable ecological end-points,
- defining the levels for those end-points that correspond to ecological goals, and
- defining the target.

In the example above, a suitable end-point for maintaining diversity might be the irreversible loss of less than 5% of all species, or the protection of 95% of the species 95% of the time in one situation, while in another situation (where there are rare or endangered species present), the end-point might be the protection of a viable population with 99% probability

The suitability of end-points will be defined by the criteria in 2.4.2. While the risk objectives may be generic for all aquatic ecosystems, the risk end-points are likely to be site-specific as they will be determined by locally important biota and habitats.

The process of setting meaningful objectives may have to involve all stakeholders. The probabilistic aspect of risk objectives is likely to be an expert process for which it is difficult to set a generic procedure. This needs further investigation.

An example of the logic that may be applied in deriving ecological objectives appears in Part 1, Appendix 1. The advantages of such a schema are that:

- it can be adjusted according to system specific knowledge;
- it is amenable to probabilistic or possibilistic expression; and
- it can be applied to assess the integrated effects of stressors (Claassen, pers. comm.; Jooste, pers. comm.).

### 3.1.2 Translating ecological risk objectives into management measures

*3.1.2.1 Transport and fate modelling*

The ERA process is initiated by the presence of a stressor source whose source profile is known. When this knowledge is combined with a knowledge of the effect of a stressor, a risk assessment can be made. The point in setting RBOs is that the stressor is known, but some characterisation of its profile is required. This process depends on the ability to model the relationship between stressor source and its environmental profile, i.e. transport and fate modelling.

Under the auspices of the US EPA much research has been done on transport and fate modelling of toxics, nutrients and system variables in aquatic systems. A number of these models are available from the Centre for Exposure Assessment Modelling of the Office of Research and Development of the USEPA (EPA, 1996). The models have been well described, and extensive literature on their use and performance is available in most cases. Examples include:

| | |
|---|---|
| Point source: | QUAL2E, PLUMES, CORMIX, EXAMS, WASP5. |
| Non-point source: | HSPF, SWAT |

Generally, the use of these models requires some expertise. A knowledge of the processes included in the model and assumptions made in their derivation is required. However, no specific modelling knowledge is required to use the models. Expertise in using these models can be gained by some application of the model by a suitably qualified person under controlled conditions. The type of training needed and the data to assess the input parameters are available and, consequently, it is felt that this is an aspect of risk for which manpower could be trained relatively easily.

*3.1.2.2 Effect modelling*

Modelling the effect of a stressor on a biotic system, while conceptually easy, is practically more difficult than transport and fate modelling. Stressor-response relationships have been determined for a number of chemicals and chemical compounds for an array of organisms and they form an important part of the aquatic toxicology literature. However, a number of problems present themselves when suitable stressor-response relationships for ecological objectives and end-points need to be selected:

- Most toxicological studies are performed on a small group of individuals from a species. Few data exist that relates to effects on a whole population, and even fewer data exists at higher levels of organisation.
- Most studies concentrate exclusively on sensitive or susceptible life stages of organisms.
- Only a few end-points are well reported on. The most popular end-points are mortality and fertility for invertebrates, and biomass density for bacteria and phytoplankton.

### 3.1.3 Stressor-response relationships

Stressor-response relationships were reviewed and discussed at a specialist workshop held during August 1998 (see Part 2 for a full workshop report). This specialist workshop arose out of the need to review the literature for information on functional relationships which may exist between selected stressors and biotic response. The following variables were selected for review (Part 2, Appendices 1-4):

- water quantity (flow)
- water quality, in the form of:
  - toxics
  - nutrients: nitrite, nitrate, ammonia, phosphate, iron, manganese.
  - system variables: pH, electrical conductivity (EC), salinity, total dissolved solids (TDS), total suspended solids (TSS), temperature.
- habitat

For the purpose of risk estimation in an ecosystem particular attention was paid to:

- source of data,
- level of organisation of the input (molecular, organism, population, community, ecosystem)
- nature of the relationship (formula/equation, expert system, etc.)
- the implicit or explicit ability to estimate or incorporate uncertainty/variability
- the level of organisation of its output

During the August workshop, flow was identified as a stressor for which much expertise exists in South Africa, due to the DWAF Instream Flow Requirements (IFRs) process. A second workshop was therefore held during January 1999, to evaluate the feasibility of using a risk-based approach to sett the water quantity for the Ecological Reserve, and when extrapolating to EMCs other than that for which the Ecological Reserve was set. It was also necessary to try and determine whether links could be made between changing flow levels and stress on riverine biota. A full workshop report can be found in Part 3.

### 3.1.4 General conclusions from the specialist workshops

*3.1.4.1 Specialist workshop 1: The use of risk-based objectives in water resource management*
This workshop introduced delegates to the concept of risk-based objectives, and discussed the feasibility of their use in water resource protection.

Discussions around stressor-response relationships highlighted the dearth of information on functional relationships. For the moment, there should be a reliance on the expert information process, particularly for flow. It was suggested that instead of developing a toxicity database for a range of South African organisms, South African data currently available should be utilized and compared to international databases. If the toxicity information is similar, international data can then be used in risk assessment. However, it is possible that mesocosm population toxicity testing needs to be undertaken, and mathematical models used to extrapolate results to field conditions.

It was accepted that the use of risk explicitly calls for setting Resource Quality Objectives, which are presently set on the basis of expert opinion, based on what constitutes a qualitative or semi-quantitative level of risk. These objectives may have to be set at a site-specific level only.

*3.1.4.2 Specialist workshop 2: Risk-based objectives flow workshop*

Most of the discussion at the workshop revolved around the approaches used for setting water quantities for Environmental Management Classes (EMCs) during Ecological Reserve assessments, and how to alter flow requirements for various EMCs. Two methods were evaluated, i.e. the 'less depth' and 'less frequency' methods, by selecting a range of rivers with a range of hydrological variability for which the necessary IFR data are available, and comparing the hydraulic and ecological consequences of applying both methods.

The results of the workshop could not show a generic preference of one method over the other for the setting of maintenance flow for different EMCs. It is possible that each case should be handled independently, the suitability of each method assessed, and the appropriate method selected. Although useful, the assessment of both methods would be costly and impractical. It was suggested that a *decision framework* be developed which could guide scientists in their assessments.

The second part of the workshop revolved around determinations of 'flow stress' and the risk of changing flows to biota. Although the original thinking was stress in terms of changing habitat, there was general recognition that IFR workshops currently proceed with implicit assessments of stress to biota. The workshop resulted in two documents, authored by Hughes and O'Keeffe respectively, for the development of a framework for determining the water quantity for determining the Ecological Reserve, and defining different levels of flow-related stress for instream riverine fauna (see Part 3). The documents suggest the use of generic stress/flow relationships combined with the natural and modified flow regimes to compare stress profiles, and where necessary modify the assurance rule curves to obtain a more balanced (in terms of stress) modified flow regime. O'Keeffe's document attempts to define a consistent qualitative description of stress levels for instream biota, which could be used to construct stress curve relationships.

There was general agreement that although these documents are very preliminary and flow-related, this approach could be followed for other stressors.

# CHAPTER 4
# CONCLUSIONS, RECOMMENDATIONS AND RESEARCH NEEDS

## 4.1 INTRODUCTION

This Chapter summarizes the main points resulting from this project, as well as the main research areas which have been identified. Some of these research areas are presently being addressed by researchers in other projects, e.g. the DWAF-CSIR collaboration to develop an ERA framework for South Africa, and some will be taken up in the second phase of the RBO research. The main point which has been emphasized during the course of this research, is the value of our fundamental knowledge of aquatic biota and ecosystems to effective water resource management. Although there is currently great emphasis on the *management* of aquatic ecosystems in South Africa, this emphasis should not undermine the value of fundamental research, which serves as the basis for effective applied management. It is imperative to increase our ecological understanding of aquatic systems before we can use the information to parameterise models, in order to account for the uncertainty and variability associated with biological information. The feasibility of using a risk-based approach to set integrated environmental objectives for South African conditions, can only be properly tested with real local ecosystem data. It is therefore necessary for scientists to articulate the value of this information, and to demonstrate how this information can be useful to users such as industry.

## 4.2 RISK MANAGEMENT AND POLICY

The use of risk-based objectives and risk assessment, although not exclusively dependent on, are certainly facilitated by risk management as a management paradigm, as opposed to hazard management (CRARM, 1996). For the effective implementation of risk procedures, the links between risk concepts, the science of risk assessment, and management and policy-making will have to be finalized and formalized. This procedure will require the development and implementation of risk policy for South Africa, at both a general and site-specific policy level. Risk communication also needs to be addressed as an important facet of risk management, as this serves to explain to users how risk management can reduce present uncertainties (stemming from a lack of knowledge) and possibly reduce over-protective measures currently in place. Risk concepts and ERA therefore need to be saleable. Both ERA and the use of risk-based objectives should be of sound scientific integrity, and should take account of political, economic and social issues; thereby requiring effective risk communication between risk managers and risk assessors (Murray and Claassen, 1999).

## 4.3 UNDERSTANDING RISK

Risk has been widely accepted as being implicit in setting Ecological Management Classes and Ecological Reserve assessments, and the two specialist workshops have generated much interest. It is now vital that scientists, practitioners and managers have a common understanding of risk and complex risk concepts. It is therefore necessary to build an awareness of risk and its role in water resource management by making it accessible and understandable.

### 4.4 RISK AND THE ECOLOGICAL RESERVE

In principle, risk-based objectives are well adapted to the needs of the Ecological Reserve, although suitable risk assessment end-points now have to be selected. To be able to set appropriate management objectives and estimate the impact of management actions, risk would need to be assessed with respect to some chosen end-point, such as the sustainability of the ecosystem. Sustainability might be set in terms of an assessment end-point such as 'the protection of 95% of all species, 95% of the time'. A policy or procedure will need to be formulated as a guideline.

The *de minimis / de manifestis* framework for risk could be used for the Ecological Reserve. The *de minimis* risk level, conceptually, is that which is considered negligible, and the *de manifestis* risk level is that which is considered unacceptable. *De manifestis* should therefore pertain to the lower (highest risk) boundary of the D management class. The actual numeric values for the *de minimis* and *de manifestis* risks need to be established to facilitate risk management.

### 4.5 INTEGRATION OF CO-OCCURRING STRESSORS

The specialist workshops detailed in Parts 2 and 3 of this document identified a number of factors which can act as stressors on aquatic biota and ecosystems, i.e. water quantity or flow, various water quality parameters, and habitat. Flow is the first variable that has been discussed in some detail, and stress/flow relationships investigated with the aim of generating stress profiles. A first attempt was also made to generate a stress index for different levels of flow-related stress for instream riverine fauna. This approach (or modifications thereof) should be followed for both water quality parameters and habitat, and expanded further for flow. The second phase of the RBO project (K8/350) will concentrate on the integration of co-occurring stressors, thereby producing an overall ecological risk. When applying risk theory to managing ecosystems, it must be recognized that stressors will rarely be present individually, but simultaneous effects on the biota will be recorded. Although the eventual effect on the ecosystem may be similar to the effect of a single stressor, the mechanisms will be quite different. The risk posed by these co-occurring stressors will therefore have to be managed accordingly.

### 4.6 RESEARCH AND DEVELOPMENT

The reviews and workshops around *stressor-response relationships* clearly showed the requirement for information regarding the causal links between stressors, the stress they place on biota, and the associated risk levels. Links between physical stream parameters, e.g. habitats, and the biota also need to be clearly understood before integrated resource quality objectives can be developed. Basic ecological information on South African biota and rivers needs to be gathered and databases developed, as this will aid understanding during the Ecological Reserve process, and when selecting ecological endpoints.

Research and development requirements therefore include the following:

- An emphasis on the collection of local basic biological and ecological data, particularly for defining stressor-response relationships. This requires a shift from data collection for the purposes of compliance monitoring, to data collection for the purpose of increasing ecological understanding.
- The production of guidelines for data collection, especially for industry. Including industry in data collection can only be successful if the *resolution of data* is specified.
- Define measurable Reserve-related end-points.
- Develop a hierarchy of stressors and understand the causal links between them, which will aid the effective management of these stressors.
- Parameterising exposure and effect models for stressors under South African conditions.
- Development of risk assessment capacity.
- Development of risk management methods and capacity, based on appropriate and effective risk management policy.
- Development of stressor-response information by developing capacity in toxicology and other testing fields.
- Investigating stressor-response relationships for selected variables, e.g. for water quality parameters, following the method used in the flow workshop.
- Further development of the concepts outlined at the flow workshop and documented in Part 3 of this report.
- Development of methods for integrating ecological risks posed by co-occurring stressors.

# CHAPTER 5
# REFERENCES


Alexander, D.G. and R. McV. Clarke (1978). The selection and limitations of phenol as a reference toxicant to detect differences in sensitivity among groups of rainbow trout (*Salmo gairdneri*). *Wat. Res.*, **12**, pp1085-1090.

Arrow K., B. Bolin, R. Costanza, P. Dasgupta, C. Folke, C.S. Holling, B. Jansson, S. Levin, K. Mäler, C. Perrings and D. Pimentel (1995). Economic growth, carrying capacity, and the environment. *Science*, **268**, pp520-521.

ASTM (1993a). ASTM D 4229-84. Conducting static acute toxicity tests on wastewaters with Daphnia. *Annual Book of ASTM Standards,* American Society for Testing Materials, Philadelphia, Pennsylvania.

ASTM (1993b). ASTM E 1193-87. Standard guide for conducting renewal life-cycle toxicity tests with *Daphnia magna. Annual Book of ASTM Standards,* American Society for Testing Materials, Philadelphia, Pennsylvania.

ASTM (1993c). Standard guide for conducting early life stage toxicity tests with fishes. *Annual Book of ASTM Standards,* American Society for Testing Materials, Philadelphia, Pennsylvania.

Bartell, S.M., R.H. Gardner and R.V. O'Niell (1992). *Ecological Risk Estimation.* Lewis Publishers, Chelsea, Michigan.

Bidwell R. (1993). *From political goals to practical application: making sustainability operational.* Proc. 1995 International Programme on the Management of Sustainability, Nijenrode, Netherlands, June 1995.

Brown, A.C. (1993). Variability in biological systems. *S. A. J. Sci.*, **89**, pp308-309.

Brown, L. (Ed.) (1993). New Shorter Oxford English Dictionary (NSOED) on historical principles. Oxford University Press, Oxford.

Bush, M.B. (1997) *Ecology of a changing planet.* Prentice Hall, New Jersey, pp434.

Chapman, C.R. and D. Morrison (1994). Impact on earth by asteroids and comets: assessing the hazard. *Nature*, **367**, pp33-39.

Chapman, G.A. (1983). Do organisms in laboratory toxicity test respond like organisms in the field? In *Aquatic Toxicology and Risk Assessment, Sixth Symposium.* W.E. Bishop, R.D. Cardwell and B.B. Heidolph (Eds.), American Society for Testing Materials, Philadelphia. Pp315-327.

Chutter, F.M. (1998). *Research on the Rapid Biological Assessment of Water Quality Impacts on Streams and Rivers.* Water Research Commission Report No. 422/1/98, Pretoria, South Africa.

Claassen, M. Personal communication. Environmentek, CSIR, Pretoria.

CRARM (1996). *Risk Assessment and Risk Management in Regulatory Decision-Making.* Commission on Risk Assessment and Risk Management. Draft Report, June 13, 1996.

Davis, J.P. (1998). *Editorial in HPS Newsletter*, **26** (11) November, 1998.

Department of Water Affairs and Forestry (1996) South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems.

Department of Water Affairs and Forestry (1997) White paper on a national water policy for South Africa.

DOE (1995). *A Guide to Risk Assessment and Risk Management for Environmental Protection.* Department of the Environment, HMSO, London, UK.

DeFinetti, B. (1990). *Theory of Probability, Volume 1.* Wiley Interscience, John Wiley and Sons, Chichester, 300pp.

EPA (1997) *Priorities for Ecological Protection: An initial List and Discussion document for EPA.* EPA/600/S-97/002, U.S. Environmental Protection Agency, Washington, DC.

Frey, H.C. (1993) Separating Variability and Uncertainty in Exposure Assessment: Motivations and Methods. *93-RA-116A.02, Air and Waste Management Association.*

Fryer, G. (1987) Quantitative and qualitative: numbers and reality in the study of living organisms. *Freshwat. Biol*, **17**, pp177-189.

Haas, C.N. (1988). Reference unknown.

HCN (1989). Assessing the risk of toxic chemicals for ecosystems. Health Council of the Netherlands, Report No 1988/28E, Den Haag, The Netherlands.

Holling, C.S. (1996) Two cultures of ecology. URL: http//www.consecol.org/vol1/iss1/art7

Jaynes, E.T. (1996) Probability Theory: The Logic of Science. URL http://www.wustl.edu.

Jooste, S. Personal communication. IWQS, DWAF, Pretoria.

King, J. and D. Louw (1998) Instream flow assessments for regulated rivers in South Africa using the Building Block Methodology. *Aq. Ecosyst. Health Manage.*, **1**, pp109-124.

Kocher, D.C. and F.O Hoffmann (1991). Regulating Environmental Carcinogens. *Environ. Sci Technol.*, **25**, pp1986-1989.

Kooijman, S.A.L.M. (1987). A safety factor for LC50 values allowing for differences in sensitivity among species. *Wat. Res.*, **21**, pp269-276.

Landis, W.R. and M-H Yu (1995). *Introduction to Environmental Toxicology: Impacts of Chemicals Upon Ecosystems.* Lewis Publishers, Boca Raton.

Mancini, J.L. (1983). A method for calculating effects, on aqautic organisms, of time varying concentrations. *Wat. Res.*, **17**, pp1355-1362.

Mayer, L.F. and M.R. Ellersieck (1986). Manual of acute toxicity: Interpretation and database for 410 chemicals and 66 species of freshwater animals. *Resource Pub. 160. U.S. Fish and Wildlife Service*, Washington, DC.

Murray, K. (1999) *National Aquatic Ecosystem Biomonitoring Programme: National Implementation Assessment.* NAEBP Report Series No 8. Institute for Water Quality Studies, Department of Water Affairs and Forestry. Pretoria, South Africa.

Murray, K. and M. Claassen (1999) An interpretation and evaluation of the US Environmental Protection Agency ecological risk assessment guidelines. *Water SA*, **25** (4), pp513-518.

National Water Act (NWA) No. 36 of 1998. *Government Gazette* **398** (19182), pp1-201.

Noss, R.F. (1990) Indicators of monitoring biodiversity: a hierarchical approach. *Conserv. Biol.*, **4** (4) pp355-364.

Palmer, C.G and P-A Scherman (in press). Application of an artificial stream system to investigate the water quality tolerances of indigenous, South African, riverine macroinvertebrates. A Report to the Water Research Commission.

Rand, G.M. (Ed.) (1995). *Fundamentals of aquatic toxicology. Effects, environmental fate, and risk assessment.* 2nd edition,Taylor and Francis, Washington, pp1125.

Roux, D.J., P.L. Kempster, C.J. Kleynhans, H.R van Vliet and H.H. du Preez (in print). Integrating Stressor and Response Monitoring into a Resource-Based Water-Quality Assessment Framework. *Environ. Management.*

Sijm, D.T.H.M., M. Schipper and A. Opperhuizen (1993). Toxicokinetics of halogenated benzenes in fish: Lethal body burden as a toxicological end point. *Environ. Toxicol and Chem.*, **12**, pp1117-1127.

Simonovic, S.P. (1996). Decision support systems for sustainable management of water resources: general principles. *Wat. Intl.*, **21**, pp223-232.

Solow R.M. (1993). Sustainability: An Economist's Perspective. In Dorfman R. and N.S. Dorfman (Eds.) *Economics of the Environment: Selected Readings*. Third Edition. WW Norton & Co, New York.

Stephan, C.E., D.I. Mount, D.J. Hanson, J.H. Gentile, G.A. Chapman and W.A. Brungs (1985). *Guidelines for Deriving National Water Quality Criteria for the protection of aquatic organisms and their uses*. U.S. Environmental Protection Agency, Duluth, Minnesota. PB85-227049.

Suter, G.W. (II) (1990). Environmental risk assessment/ environmental hazard assessment: Similarities and differences. In: *Aquatic Toxicology and Risk Assessment: Thirteenth Volume*. W.G. Landis and W.H. van der Schalie (eds.), ASTM STP 1096, American Society for Testing Materials, Philadelphia, pp5-15.

Suter, G.W. (II) (Ed.) (1993). Ecological risk assessment. Lewis Publishers, Michigan. pp538.

Travis, C.C., S.A. Richter, A.C. Crouch, R. Wilson and E.D. Klema (1987). Cancer Risk Management. *Environ. Sci. Technol.*, **21**, pp415-420.

Tucker, R.K. and J.S. Lietzke (1979). Comparative toxicity of insecticides for vertebrate wildlife and fish. *Pharmac. Ther.*, **6**, pp167-220.

USEPA (1992). *Framework for Ecological Risk Assessment*. Risk Assessment Forum, United States Environmental Protection Agency, Washington DC. EPA/630/R-92/001.

USEPA (1996). *Exposure Models Library and Integrated Model Evaluation System*. United States Environmental Protection Agency, Office of Research and Development EPA/600/C-92/002 Revised March, 1996.

USEPA (1998). *Guideline for Ecological Risk Assessment*, Risk Assessment Forum, US Environmental Protection Agency, Washington, DC. Document EPA/630/R-98/.

USEPA (1998). ECOTOX data base system. URL: http:// www.epa.gov/ecotox/.

Walters C.J. (1986). *Adaptive Management of Renewable Resources*. Macmillan, New York.

Whipple, C (Ed.) (1987). *De Minimis Risk*. Plenum Press, New York.

## APPENDIX 1
## AN INFERENCE SCHEMA FOR ECOLOGICAL RESERVE RISK ESTIMATION

The conceptual model can be summarised as follows:
It is assumed that ecosystem sustainability will only be impaired if ecosystem integrity is impaired, and this will only take place if key populations become extinct. The extinction of key populations can be due to either unsustainably low population numbers or massive mortality or both. Unsustainably low numbers of key populations can be due to either or both effects of chronic exposure or disrupted biotic interactions (such as by the introduction of foreign species or over-harvesting of other prey species). Massive mortality presupposes the occurrence of acute stress. Chronic stress occurs when either or all of a number of stressors occur at chronic levels. While acute stress occurs when any or all of the stressors occur at acute levels. This conceptual model can be formalised in the following inference schema:

- IF (ecosystem sustainability is impaired) THEN (ecosystem integrity is compromised).
- IF (ecosystem integrity is compromised) THEN (key populations is extinct)
- IF (key populations is extinct) THEN (population numbers are too low) OR (massive mortality occurs)
- IF (population numbers are too low) THEN (chronic effects occur) OR (biotic interactions are disrupted)
- IF (massive mortality occurs) THEN (acute stress occurs)
- IF (chronic effects occur) THEN (chronic levels of stressor 1 occurs) OR (chronic levels of stressor 2 occurs) OR ....
- IF (acute stress occurs) THEN (acute levels of stressor 1 occurs) OR (acute levels of stressor 2 occurs) OR...

To simplify, the following key to events is used: ecosystem sustainability is impaired = S, ecosystem integrity is compromised = I, key populations is extinct = K, population numbers are too low = N, massive mortality occurs = M, chronic effects occur = C, chronic levels of stressor *I* occurs = $X_i$, acute stress occurs = A, acute levels of stressor *I* occurs = $Y_i$.

For each statement there is a specific expectation of it being true as well as an expectation that each event is true. There are then two ways of approaching the estimation of Ecological Reserve risk:
Ordinary conditional expectation can be used to estimate the expectation of A if the expectation of B is given. This is then read as "The probability of event A given that event B occurred". It can be calculated by the rules of probability theory as: P(A/B)* P(B) = P(A and B). Then:

- $P(S \cap I) = P(S/I)*P(I)$
- $P(I \cap K) = P(I/K)*P(K)$
- $P(K) = P(K/N \cup M)$
- $P(N) = P(N/C \cup B)$
- $P(M)=P(A)$
- $P(C) = P(C/X_1 \cup X_2 \cup ... \cup X_n)$
- $P(A) = P(A/Y_1 \cup Y_2 \cup ... \cup Y_n)$

where

$P(A)$ : Expectation expressed as: the probability of A.

$P(A|B)$ : The probability that event A occurs given that event B has occurred.

$P(A \cap B)$ : The probability that events A and B both occur. According to probability theory $P(A \cap B) = P(A|B)*P(B) = P(B|A)*P(A)$

$A \cap B$ : Given sets A and B, $A \cap B$ is the intersection between the sets. This designates the elements of A that are also elments of B. It is used here as being equivalent to the logical disjunction, i.e. if A and B are propositions, then $A \cap B$ designates A AND B are simultaneously valid.

$A \cup B$ : Given sets A and B, $A \cup B$ is the union of the two sets. In the sense used here it is equivalent to the logical disjunction, i.e. if A and B are propositions then $A \cup B$ indicates that A OR B (not exclusively) are valid.

Some of these statements can be assumed to take the form: "If, and only if (A) then (B)". It is a very strong statement to say that the consequent and precedent in the statement are necessary and sufficient preconditions of each other, but this might be done as a matter of policy to achieve specific goals such as in a precautionary approach. Then the conditional probability $P(A/B) = 1$. If it is assumed that ecosystem sustainability will be impaired if and only if ecosystem integrity is compromised and that ecosystem integrity will be compromised if and only if key species become extinct.

Assuming this model describes the ecosystem level interactions would imply that from a knowledge of the characteristics of the occurrence of stressors, it is possible to:

- infer the expectation of sustainability and
- highlight the knowledge gaps for the system being assessed.

## APPENDIX 2
## A GENERIC PROCEDURE FOR ECOLOGICAL RISK ASSESSMENT

The following generic procedure for carrying out an Ecological Risk Assessment was taken from Suter (1993).

A) Choose endpoints:
   1) What valued components of the ecosystem are at risk? Set:
      a) matrix of component interactions potentially affected
      b) list organisms most exposed (receptor identification)
      c) indirect effects tree of linkages
      d) review species sensitivity
   2) Define endpoints operationally
      - must contain: subject (e.g. species x) + characteristic (e.g. significant inhibition of reproduction)
   3) Translate into numeric form
      a) make decision on significance of effects and set acceptable threshold (e.g. 10% reduction in reproduction) AND/OR
      b) calculate % probability of exceeding a preselected threshold, OR
      c) express effects as function of probability and magnitude

B) Describe environment
   1) Conceptualise environment
      - chose reference environment (e.g. worst case, reasonable worst case, most likely case)
   2) Define boundaries of the environment:
      - legal (e.g. national/international) or functional (e.g. catchment) OR
      - decide where the chosen threshold concentration boundary would be

C) Obtain source terms:
   1) Spatial scale of release
   2) Temporal scale of release

D) Assess exposure by:
   1) Assessing routes of exposure
   2) Modelling relevant receiving environment in terms of:
      a) transport
      b) dilution
      c) transformation
      d) degradation
      e) partitioning
   3) Assessing natural background

E) Assess effects through:
   1) Ecological epidemiology (field observation of effects) OR
   2) Toxicity testing, paying particular attention to:
      a) the effect of the combination of level and duration of exposure on responses of concern

b) - the statistical model fitted to the toxicity data OR
- the "exposure-response" or other numerical summary of data chosen

c) the effect models generated reflecting assumption on nature of relationship between test endpoint and assessment endpoint (ranging from "the test reflects all relevant features of response" to mathematical models simulating mediation of toxicity responses by population and ecosystem processes)

d) derivation of a model function relating " level of effect on assessment endpoint" to exposure, by parameterising model from test data and ecosystem processes

F) Characterise the risk, by:

1) Selecting the correct dimensionality of exposure and effect data (e.g. is duration (time) a factor, could concentration and time be collapsed into exposure [conc*time], should "proportion responding" and "severity of response" be collapsed into "mortality")
2) Scale data appropriately (e.g. total concentration in medium, dissolved concentration, total body burden etc.)
3) Select the appropriate model dimensionality for the temporal and spatial extent of the assessment
4) Derive risk estimate

a) derive joint probability of exposure and effect (area of overlap between functions) OR

b) calculate risk at e.g. most plausible, best and worst cases (say, 50th 5th and 95th percentile)

c) prepare narrative characterisation, containing:

i description of models (assumptions, validity, peer acceptance)
ii sources and quality of input data
iii sources of uncertainty and their quantification
iv context for assessment results
- conflicting evidence and its explanation
- alternate credible assumptions and explanations
- research to resolve major uncertainties
- precedent assessments and analogous situations

d) distinguish between *de manifestis* and *de minimis* risk (clearly unacceptable and clearly trivial risk)

# PART 2

## SPECIALIST WORKSHOP 1:

## THE USE OF RISK-BASED OBJECTIVES IN WATER RESOURCE MANAGEMENT

This Part contains the output of an introductory workshop to risk and risk-based concepts, and discussion around the feasibility of this approach for water resource management. Stressor-response relationships were introduced and discussed in terms of water quantity, water quality and habitat.

## TABLE OF CONTENTS

## INTRODUCTION

An important component of utilizing risk-based objectives (RBO) for the protection of water resources is to identify impacts from multiple physical and chemical parameters on biological systems, evaluate the relative importance of each factor, and then attempt to integrate the impacts on the biota. This approach, known as *ecological risk analysis*, can be used to assess the probability and magnitude of adverse effects that could arise from exposure to single or multiple stressors. When attempting to determine the overall ecological risk to an aquatic system, a number of factors must be considered. These include identifying stressors, exposure assessments i.e. estimating the level of exposure of the aquatic community to chemical and/or physical habitat stressors, ecological receptor characterization, and ecological effects characterization. Risk characterization and risk management complete the process (Paulson and Parkhurst, 19xx).

Central to ecological risk analysis is the *stressor-response relationship*, which is used to estimate the probability of effect. This specialist workshop arose out of the necessity to review the literature for information on functional relationships which may exist between selected stressors and biotic response. In other words, do functional relationships exist that relate the occurrence of stress to an observable biotic effect? and what are the uncertainties involved in using this relationship? With the exclusion of toxics, little information is available in South Africa on stressor-response relationships, and the international literature was reviewed on a limited scale. This information was subsequently presented at the specialist workshop for discussion.

The following variables were selected for review, as they form the components of the Resource Quality Objectives which would have to be set in terms of risk:

- water quantity (flow)
- water quality, in the form of:
  - toxics
  - nutrients - nitrite, nitrate, ammonia, phosphate, iron, manganese.
  - system variables - pH, electrical conductivity (EC), salinity, total dissolved solids (TDS), total suspended solids (TSS), temperature.
- habitat

*Note: Toxics were exhaustively reviewed for the water quality guideline project; another review was therefore not undertaken. No stressor-response information could be found for TSS in the limited time available.*

Literature reviews were conducted to assess the availability of the following information about stressor-response relationships:

- Input to the relationship i.e. what is the stressor and what is its metric?
- How are the inputs generated, and what level of inputs are needed?
- What are the parameters in the relationship, e.g. what prior knowledge is needed or experimental condition maintained?
- Can the relationship be described as an equation?
- What sort of output does the relationship produce, and what are their metrics?
- Can the relationship predict output uncertainty from input uncertainty?

- What are the limitations on the use of the relationship, e.g. site-specific, species-specific?

Results of the literature reviews are shown in the tables of Part 2, Appendices 1-4.

The workshop was attended by the project team and the following delegates:

Joseph Matjila, IWQS
Toni Belcher, IWQS
Prof Denis Hughes, IWR
Ms Delana Louw, IWR Environmental
Mr Drew Birkhead, Streamflow Solutions
Mr Dez Weeks, IWR
Mr Dirk Roux, Environmentek, CSIR
Mr Marius Claassen, Environmentek, CSIR
Dr Neels Kleynhans, IWQS
Mr Gareth McKonkey, DWAF

**WORKSHOP OBJECTIVES**

- Introduce delegates to the concept of risk-based objectives, and discuss the feasibility of their use in water resource protection.
- Introduce and discuss risk-based related issues, e.g. risk, the Reserve and the classification framework; integrated risk concepts; risk assessment initiatives such as the RBO project; and technology needs.
- Discuss quantifying variables in terms of risk.
- Discuss stressor-response relationships.

**WORKSHOP OUTCOMES**

***Setting Resource Quality Objectives in terms of risk***

Setting Ecological Management Classes (EMCs) for the Reserve is based on two assumptions which incorporate risk concepts. Assumption 1 is that measurable Resource Objectives for maintaining resource integrity, can be related to the risk of using or affecting the Resource Base. The second assumption is that from chemistry and habitat conditions one can quantify the conditions that will allow given levels of integrity to be maintained in the long term, even if disturbances occur. It is however necessary to recognize three kinds of risk: 1) the risk of utilizing the Resource Base, 2) the risk associated with not achieving the guideline value, e.g. the closer you get to the guideline value for a particular variable, the greater the risk of affecting the system, and 3) risk to the biota.

Despite the research difficulties associated with setting Resource Quality Objectives in terms of risk, the link between the science of risk and management is crucial. The US EPA use risk in three categories: 1) effluent standard, 2) effluent criterion (whole effluent toxicity (WET) testing), and 3) in-stream biomonitoring (river health surveys). While the process in the US is risk-based, the final auditing for regulatory purposes is hazard-based. The procedures to be followed in South Africa would have to be clearly defined.

***Risk calculation or estimation procedures e.g. flow***

Due to the wealth of information available from the In-stream Flow Requirement (IFR) process, an attempt was made to discuss the use of risk in setting flows for EMCs in quantity assessments for the Ecological Reserve.

During the IFR process a set of rules are set up according to which flows are managed. These rules already have a probabilistic function. To set flows per EMC, maintenance flows must be set per river. The question is how to link setting maintenance flows per river to EMCs; possibly based on % assurances or depth. It is possible to set four curves to determine flows per river, for the four categories of risk i.e. EMCs. These will have to be site-specific and developed per river, as four different maintenance flows will have to be set per river.

Recommendations or requirements for flow are therefore as follows:

- IFR teams should start thinking in terms of risk.
- Setting up risk curves will always be site-specific and dependent on expert judgement.
- A nationwide database of information from regional experts will be required before risk curves can be developed.
- A large database reduces uncertainty, but requires more time to set up.

## WORKSHOP CONCLUSIONS BASED ON STRESSOR-RESPONSE TABLES (APPENDICES 1-4)

- Very little South African flow information exists for stressor-response relationships. For the moment, there should be a reliance on the expert information process.
- Instead of developing a South African toxicity database for many organisms, rather utilize South African data and compare to international databases. If the toxicity information is similar, international data can then be used.
- Acute toxicity information (e.g. LC50s: that concentration of a stressor responsible for the death of 50% of the test population) is not very useful. Long-term effects and LT50s (the time needed for the death of 50% of the test population ) are more useful.
- Ecosystem-type (mesocosm) population toxicity testing needs to be undertaken, and mathematical models used to extrapolate results to field conditions. Exposure models are available for variables such as toxics, nutrients, TDS, DO and pH.

## FINAL CONCLUSIONS

- EMCs can have associated levels of risk of change to the Resource Base.
- Risk can be defined as damage to the Resource Base, and is related to ecological integrity and diversity.
- Resource Quality Objectives are presently set on the basis of expert opinion, based on what constitutes a qualitative or semi-quantitative level of risk.
- The use of risk explicitly calls for setting Resource Quality Objectives. However, these objectives may have to be site-specific only.

## REFERENCES

Paulson, C. and Parkhurst, B. (19xx). Application of aquatic ecological risk methodology to support site-specific water quality-based permitting. Reference unknown.

# APPENDIX 1
# LITERATURE REVIEW ON STRESSOR-RESPONSE RELATIONSHIPS - FLOW:
*Derek Weston, IWQS*

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (metric) | |
| Lab | Fish species (Roach and Dace) | Flow ( critical velocity or displacement velocity | Laminar flow | Regression equation | Variance | Population | Population growth (individuals present) | Mann and Bass (1997) |
| Field | Aquatic ecosystem | Flow (reduction velocity) | Alpine conditions | Rules | | Community | Population (change in make up) | Bundi, Eichenberger and Peter (1990) |
| Field | Fish species (chinook Salmon) | Habitat Capacity, min. instream flows( presence or absence) | US environment | Equation | | Fish species | Population (growth) | Cardwell, Jager and Sale (1996) |
| Field | Fish species (Sturgeon) | Flow (volume) | Russian environment | Curves | | Fish species | Population (productivity) | Fachchevsky (1994) |
| Field | Community (flood plain grass) | Maximum spring floods (probability) | Russian environment | Curves | | Community | Population (productivity | Fachchevsky (1994) |
| Model | Fish species | Taxonomic unit (presence or absence) | Temperature | Rules | | Ecosystem | Population (catch per unit effort) | Jewitt, Weeks, Heritage, van Niekirk and O'Keefe (1997) |
| Field | Fish species | Habitat area (square metres) | Sampling and US environs | Regression equation (graphs) | | Ecosystem | Population (species richness) | Schlosser (1985) |
| Previous studies | Aquatic ecosystem | Habitat area (square metres) | Sampling and US environs | PHABSIM model | Possible | Ecosystem | Habitat (relationship with streamflow) | Douglas and Johnson (1991) |
| Previous studies | Aquatic ecosystem | Flow (rapid variability) | US environs | General concepts (could possibly make rules) | | Ecosystem | Population (presence or absence) | Cushman (1985) |
| Lab | Species (tadpoles) | Flow (velocity) | Temperature | Rule | | Species population | Population (individuals/time) | Odendaal and Bull (1980) |
| Field | Macro invertebrates | Flow (velocity) | Reach scale hydraulic conditions | Rule | | Invert community | Population (individuals/time) | Lancaster and Hildrew (1993) |
| Field | Ecosystem | Flow (variability) | New Zealand environs | General rules | | Ecosystem | Conditions (favourability) | Biggs et al (1990) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (metric) | |
| Lab | Fish species (Roach and Dace) | Flow ( critical velocity or displacement velocity | Laminar flow | Regression equation | Variance | Population | Population growth (individuals present) | Mann and Bass (1997) |
| Field | Ecosystem | Flow (variability) | New Zealand environs | Statistical analysis | | Differing communities | Conditions (univariate analysis of variance) | Jowett and Duncan (1990) |
| Field | Benthic invertebrates | Flow (velocity and change) | | General observations (rules) | | Benthic inverts (species) | Populations (drift densities) | Irvine and Henriques (1984) |
| Field | Ecosystem | Flow (% of virgin flow conditions) | | Rule | | Ecosystem | Populations (biotic impact) | O'Keefe and Davies (1990) |
| Previous studies | Ecosystem | Flow | | Equation | | Ecosystem | Integrity | Amir and Hyman (1993) |
| Field | Fish species (9) | Flow (discharge -weighted usable area) | Northern hemisphere fish | Curves and optimum discharge equations | | Weighted usable area and biomass | Optimum flow | Orth and Leonard (1990) |
| Field | Macro invertebrates | Flow (habitat area and volumes) | Australian species | PHABSIM model | | Macro invertebrates | Conditions(Habitat area and discharge) | Gippel et al (1996) |
| Field | Fish species | Flow (habitat area, depths and velocities) | Australian species | PHABSIM model | | Fish species | Conditions ( velocity and depth) | Stewardson, Gippel and O'Neill (1996) |
| Field | Benthic macroinverts | Flow (volume) | Cape perennial rivers | uncertain | uncertain | Benthic communities | Indices of flow | Tharme (1996) Abstract only |
| Field/lab | Benthic and fish communities | Flow (volume and weighted usable area) | | PHABSIM model | | Ecosystem | Habitat (minimum required flow) | Gore and King (1989) |
| Field | Macrophytes | Flow (velocity) | | Curve regression | | Macrophyte communities | Habitat (Volume as a % of cross sectional area) | Biggs (1996) |
| Model | Fish species | Flow (velocity and depth) | | Curves | | Fish species | Habitat suitability (index) | Heggenes (1996) |
| | Fish species (Brown Trout) | Flow (velocity) | | Curves | | Population | Population (cumulative frequency) | Heggenes (1996) |
| Field | Benthic macro invertebrates | Flow (velocity and depth) | | Charts and unimodal response curves | | Benthic macroinvertebrates | Population (density in numbers per $m^2$) | Fjellheim (1996) |
| Field | Fish species and benthic macro invertebrates | Flow (habitat, depth and velocity) | | PHABSIM model, preference curves | | Fish species and benthic macro invertebrates | Habitat integrity (suitability index) | King and Tharme (1994) |

## GENERAL NOTES

- Magnitude of change is not necessarily directly related to the magnitude of the physical change.
- Some degree of disturbance is necessary to maintain diversity of natural communities.
- There is generally a lag period before ecological changes are noticed.
- Cannot predict the consequence of changing flow conditions to biota without knowing what will happen to the habitat (including the form and position of the river). Focus should first be on water for the habitat.
- There is difficulty in taking results form laboratory experiments and applying them to field conditions, due to the role of habitat. For example, a critical flow velocity may displace larval fish, however, in the field the fish will take cover in reeds and macrophytes.
- Many of the studies focus on direct impacts of flow and do not study the indirect impacts such as the structure of the river bed, sedimentation, chemical conditions, temperature etc.
- "The literature on the specific ecological effects of hydrological regime on stream ecosystem structure and function is incomplete, largely because hydrological regime is an integrative descriptor of numerous selected forces and habitat conditions, particularly extreme events, play a central role in stream ecology and that climate change that alters these conditions has important implications for stream processes and patterns" (Poff, Tokar and Johnson,1996).
- Refugia have to be considered.
- Very little South African flow information exists for stressor-response relationships. For the moment, there should be a reliance on the IFR expert information process.

**APPENDIX 2**
**LITERATURE REVIEW ON STRESSOR-RESPONSE RELATIONSHIPS - NUTRIENTS:**
*Brendan Hohls, IWQS*

## NITRITE

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncertainty calculation | Level | Type (metric) | (Other) | |
| Lab | Species *Micropterus treculi* | Exposure (Conc., time) | temperature, exposure time to nitrite | 96 hr LC50: 187.6 ± 12.1 $mg.l^{-1}$ nitrite | Yes | organism | survival (50 %) | Nitrite 1 (see attached) | Tomasso *et al.* (1986) |
| Lab | Hybrid species<br>Sunshine bass (1 - 2 g) | Exposure (Conc., time) | temperature (9 to 25 C), exposure time to nitrite, salinity | 96-hr LC50: 12.8 ± 1.6 $mg.l^{-1}$ nitrite | Yes | organism | survival (50 %) | Nitrite 2 | Weirich *et al.* (1993) |
| Lab ? | Species *Ictalurus punctatus* | Exposure (Conc., time) | exposure time to nitrite | 96-hr LC50: 7.1 $mg.l^{-1}$ nitrite (with high chloride) | No | organism | survival (50 %) | Nitrite 3 | Dallas and Day (1993) |
| Lab ? | Species - Rainbow trout and others | Exposure (Conc., time) | exposure time to nitrite, nitrite, temperature, pH | 72- & 96-hr LC50 | Yes (confidence limits) | organism | survival (50 %) | Nitrite 4 | Russo and Thurston (1977) |
| Lab | Species *Clarias lazera* | Exposure (Conc., time) | exposure time to nitrite, fish weight | 96-hr $TL_m$ ÷ LC50 | No | organism | median tolerance ÷ survival (50 %) | Nitrite 5 | Hilmy *et al.* (1987) |
| Lab | Species Fathead minnows and others | Exposure (Conc., time) | exposure time to nitrite, fish weight, temperature | 96-hr LC50: 150 ± 5 $mg.l^{-1}$ nitrite | Yes | organism | survival (50 %) | Nitrite 6 | Tomasso (1986) |

**NITRATE**

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert. calculation | Level | Type (metric) | | |
| Lab | Species *Micropterus treculi* | Exposure (Conc., time) | temperature, exposure time to nitrate | 96 hr LC50: 1261 ± 142.1 $mg.l^{-1}$ nitrate | Yes | organism | survival | | Tomasso *et al.* (1986) |

**AMMONIA**

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert. calc. | Level | Type (metric) | | |
| Lab | Species *Micropterus treculi* fingerlings | Exposure (Conc., time) | temperature (22 °C), exposure time to ammonia, pH (8) | 96-hr LC50: 12.7 ± 0.9 $mg.l^{-1}$ total ammonia 0.56 $mg.l^{-1}$ UIA | Yes | organism | survival (50 %) | Ammonia 1 | Tomasso *et al.* (1986) |
| Lab | Hybrid species Sunshine bass | Exposure (Conc., time) | temperature (9 to 25 °C), exposure time to ammonia, pH | 96-hr LC50: 0.70 ± 0.04 $mg.l^{-1}$ unionised ammonia (UIA) | Yes | organism | survival (50 %) | | Weirich *et al.* (1993) |
| Lab ? | Species *Lepomis cyanellus* | Exposure (Conc., time) | exposure time to ammonia | 96-hr LC50: 0.89 $mg.l^{-1}$ UIA | No | organism | survival (50 %) | Ammonia 2 | Dallas and Day (1993) |
| Artificial stream | Species of fish and macroinvert. communities | Exposure (Conc., time) | Mean temp., mean pH, exposure time to ammonia | Highest $NH_3$ no-effect and lowest effect concentrations | Range | organism | growth effects and standing stock | Ammonia 3 | Hermanutz *et al.* (1987) |
| Lab | Species (*Daphnia* and some fish) | Exposure (Conc., time) | temperature (23 ± 2 °C), pH (7.49 - 8.34), exposure time to ammonia | 48-hr LC50: Daphnia :- 0.39 $mg.l^{-1}$ UIA | No | organism | survival (50 %) | Ammonia 4 | Gulyás and Fleit (1990) |
| Lab (with | Species of | Exposure | temperature, pH, | 96-hr LC50 | Range | organism | survival (50 %) | Ammonia 5 | Arthur *et al.* |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert. calc. | Level | Type (metric) | | |
| river water) | invertebrates and fish | (Conc., time) | exposure to ammonia | | | | | | (1987) |
| Lab static test | Species of New Zealand fish and invertebrate | Exposure (Conc., time) | temperature (15°C), pH, exposure to ammonia | 96-hr LC50 | Yes | organism | survival (50 %) | Ammonia 6 | Richardson (1997) |
| Lab static test | Species of striped and hybrid bass | Exposure (Conc., time) | temperature (20 to 22 °C), pH (8.2-8.4), exposure to ammonia | 96-hr LC50 | Yes | Organism | survival (50 %) | Ammonia 7 | Oppenborn and Goudie (1993) |

**PHOSPHATE**

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert. calc. | Level | Type (metric) | | |
| Lab | Species *Ictalurus punctatus* | Exposure (Conc., time) | Temp., exposure to phosphate | cardiac and opercular rate observations | No | organism | % increase over control | Phosphate 1 | Strange *et al.* (1978) |

**Phosphate data**

Phosphate 1:

Peak mean cardio-opercular rates occurred at 12 mg.$l^{-1}$ phosphate. At the end of the study, both the cardiac and opercular activity rates returned to their pre-experiment levels. It was further found that high values of phosphate do not further alter the cardiac and opercular rates. Phosphate stress is momentary, and that removal of the stress allowed the catfish to return to basal cardiac rhythm.

**IRON**

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert. calc. | Level | Type (metric) | | |
| Lab - flow-through tank | Species *Tilapia sparrmani* | Exposure (Conc., time) | Temperature, exposure to iron, pH | Observation only | No | organism | No. of mortalities (unclear total number tested) | Iron 1 | Wepener *et al.* (1992) |
| Lab | Species *Tilapia sparrmani* | Exposure (Conc., time) | Temperature, exposure to iron | Observation only | No | organism | Increase in fish activity and gill damage | Iron 2 | Grobler *et al.* (1989) |

**Iron data**

Iron 1:
Iron and manganese effects on hematology were the focus of the study, not mortality, therefore, mortality was not well reported. During exposure to Fe (1.57 $mg.l^{-1}$ $FeCl_3$), 9 mortalities (out of a minimum of 10 test fish - uncertain) were recorded at pH 5 and 3 mortalities were recorded at pH 7.4. The bioassay lasted for 96 hours. It was, however, stated that any external stressor, even those which are not considered lethal, can have a detrimental effect on aquatic organisms.

Iron 2:
Addition of iron (88 $mg.l^{-1}$ Fe) resulted in a yellowish colour and an increase in turbidity, which resulted in an increase in fish activity. Increased metabolism due to increased activity caused by stress led to a drastic increase in oxygen consumption. A scanning electron micrograph of gill tissue after exposure to sublethal iron concentrations for 72 hours revealed a collapse of the gills as well as an increase in the number of mucous cells.

### MANGANESE

| INPUT | | | RELATIONSHIP | | | OUTPUT | | DATA | REF |
|---|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (metric) | Parameters | Form | Uncert.. calc. | Level | Type (metric) | | |
| Lab - flow-through tank | Species *Tilapia sparrmanii* | Exposure (Conc., time) | Temperature, exposure to manganese, pH | Observation only | No | organism | No. of mortalities (unclear total number tested) | Manganese 1 | Wepener *et al.* (1992) |

**Manganese data**

Manganese 1:
Iron and manganese effects on hematology were the focus of the study, not mortality, therefore, mortality was not well reported.
During exposure to Mn (4.43 $mg.l^{-1}$ manganese chloride), no mortalities were recorded at pH 5 or pH 7.4. The bioassay lasted for 96 hours. It was, however, stated that any external stressor, even those which are not considered lethal, can have a detrimental effect on aquatic organisms.

### GENERAL NOTES

- Most of the stressor-response information is northern hemisphere toxicity data at the organism level.
- We should however be looking at *trophic status* or *enrichment*, not toxicity effects.

### REFERENCES

Dallas, H.F. and Day, J.A. (1993). The effect of water quality variables on riverine ecosystems: a review. WRC TT 61/93. p vii, 25, 79 - 91.

Weirich, C.R.; Tomasso, J.R. and Smith, T.I.J. (1993). Toxicity of ammonia and nitrite to sunshine bass in selected environments. Journal of Aquatic Animal Health, Vol. 5. p 64 - 72.

Tomasso, J.R. and Carmichael, G.J. (1986). Acute toxicity of ammonia, nitrite and nitrate to the Guadalupe bass, *Micropterus treculi*. Bulletin of Environmental Contamination and Toxicology, Vol. 36. p 866 - 870.

Hermanutz, R.O.; Hedtke, S.F.; Arthur, J.W.; Andrew, R.W.; Allen, K.N. and Helgen, J.C. (1987). Ammonia effects on macroinvertebrates and fish in outdoor experimental streams. Environmental Pollution, Vol. 47, no. 4. p 249 - 283.

Gulyás, P. and Fleit, E. (1990). Evaluation of ammonia toxicity on *Daphnia magna* and some fish species. Aquacultura Hungarica (Szarvas), Vol. 6. p 171 - 183.

Arthur, J.W.; West, C.W.; Allen, K.N. and Hedtke, S.F. (1987). Seasonal toxicity of ammonia to five fish and nine invertebrate species. Bulletin of Environmental Contamination and Toxicology, Vol. 38. p 324 - 331.

Richardson, J. (1987). Acute ammonia toxicity for eight New Zealand indigenous freshwater species. New Zealand Journal of Marine and Freshwater Research, Vol. 31. p 185 - 190.

Oppenborn, J.B. and Goudie, C.A. (1993). Acute and sublethal effects of ammonia on striped bass and hybrid striped bass. Journal of the World Aquaculture Society, Vol. 24, no. 1. p 90 - 101.

Hoffman, D.J.; Rattner, B.A.; Burton, G.A. Jnr. and Cairns, J. Jnr. (1995). Handbook of Ecotoxicology. CRC Press, Inc., United States of America. p 519 - 520.

DWAF (1996). South African Water Quality Guidelines, Volume 7: Aquatic Ecosystems. First Edition. Department of Water Affairs and Forestry.

Schoore, J.E.; Simco, B.A. and Davis, K.B. (1995). Responses of blue catfish and channel catfish to environmental nitrite. Journal od Aquatic Animal Health, Vol. 7. p 304 - 311.

Russo, R.C. and Thurston, R.V. (1977). The acute toxicity of nitrite to fishes. USA Environmental Protection Agency Ecological Research Series. EPA -600/3-77-085. p 118 - 131.

Hilmy, A.M.; El-Domiaty, N.A. and Wershana, K. (1987). Acute and chronic toxicity of nitrite to *Clarias lazera*. Comp. Biochem. Physiol., Vol. 86C, no.2. p 247 - 253.

Tomasso, J.R. (1986). Comparative toxicity of nitrite to freshwater fishes. Aquatic Toxicology, Vol. 8. p 129 - 137.

Jensen, F.B. (1996). Uptake, elimination and effects of nitrite and nitrate in freshwater crayfish (*Astacus astacus*). Aquatic Toxicology, Vol. 34. p 95 - 104.

Strange, J.R.; Fletcher, D.J. and Allred, P.M. (1978). Cardiac and opercular responses in channel catfish (*Ictalurus punctatus*) exposed to high levels of phosphate. J. Environ. Sci. Health, A13(7). p 501 - 515.

Wepener, V.; Van Vuren, J.H.J. and Du Preez, H.H. (1992). Effect of manganese and iron at a neutral and acidic pH on the hematology of the banded tilapia (*Tilapia sparrmanii*). Bulletin of Environmental Contamination and Toxicology, Vol. 49. p 613 - 619.

Grobler, E.; Du Preez, H.H. and Van Vuren, J.H.J. (1989). Toxic effects of zinc and iron on the routine oxygen consumption of *Tilapia sparrmanii* (Cichlidae). Comp. Biochem. Physiol., Vol. 94C, no.1. p 207 - 214.

Mehner, T. and Benndorf, J. (1995). Eutrophication - a summary of observed effects and possible solutions. J. Water SRT - Aqua, Vol. 44. p 35 - 44.

Ufodike, E.B.C. and Onusiriuka, B.C. (1990). Acute toxicity of inorganic fertilizers to African catfish *Clarias gariepinus* (Tengals). Aquaculture and Fisheries Management, Vol.21. p 181 - 185.

# APPENDIX 3
## LITERATURE REVIEW ON STRESSOR-RESPONSE RELATIONSHIPS - SYSTEM VARIABLES:

*Patsy Scherman and Nikite Muller, IWR*

**pH** *(SCHERMAN)*

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 1. Laboratory | Cellular - cells of green alga *C. reinhardtii* | Exposure (uptake of selenate + selenite, pH) | Culture medium, species, cell density, incubation time | | No | Organism | $^{75}Se$ uptake ($^{75}Se$ concen / cell mass in 6 hrs) |
| 2. Laboratory | Species - water mite *Arrenurus manubriator* | Exposure (hatching + survival, pH) | Soft water, static renewal system, photoperiod, temperature, incubation time | Logistic response function | No | Organism | Hatching + survival i.e. number / pH reading (probability) |
| 3. Laboratory | Species -aquatic midge *Chironomus riparius*. | Ditto | Ditto | Ditto | No | Organism | Ditto |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 4. Laboratory | Species - bacterium *Sphingomonas* sp. strain P5 | Exposure (growth rate on PCP + biodegradation rate of PCP, medium pH) | Nutristat culture (i.e. continuous culture at a controlled substrate conc.), pentachlorophenol (PCP) as C + energy source, temperature | - Cardinal pH model<br><br>EC50 at pH 7.1=830$\mu$M PCP.<br><br>- Biodegradation curve<br><br>Non-linear regression analysis based on the algorithm of Marquardt | No | Organism | - Growth rate on PCP (mass of bacterial culture<br><br>- Biodegradation rate of PCP by bacteria (PCP concen. (measured by absorbance + HPLC) |
| 5. Field | Benthic invertebrate community structure | Taxonomic unit (presence/absence + density, pH) | Sampling regime | Rule base | No | Aquatic ecosystem | Ecological integrity (i.e. highest no. of genera at lowest pH's) |
| 6. Field | Macroinvertebrate species assemblages + taxon richness | Taxonomic unit (correlation with various pH levels) | Sampling regime (riffle-dwellers) | Spatial patterns as shown by ordination (Decorana), classification (Twinspan) + multiple disciminant analysis | No | Taxonomic unit as indicator of acid stream conditions | Change in taxonomic richness (integrity) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 7. Laboratory - outdoor channels | Species - fathead minnow *Pimephales promelas*, + macroinvertebrate community | Exposure (densities / numbers / behavioural responses, pH 8 (ambient) vs pH 6 vs pH 5 | Fish species, invertebrate sampling regime, experimental channels, Mississippi River water, 17 weeks | Kruskal-Wallis + ANOVA | No | Population (fish) + community (macroin-vertebrates) | Altered macro-invertebrate community structure (diversity index)<br><br>Altered behavioural responses (e.g. drift, emergence) per taxon<br><br>Altered reproduction in fathead minnows (spawning / parent)<br><br>Reduced development (eggs to minnow eyed-embryos) |
| 8. Laboratory - outdoor (semi-natural) channels | Macroinvertebrate lotic community (natural colonization) | Exposure (total abundance, pH) | Macroinvertebrate lotic community, plasticized wooden channels, (Al), 3 months, constant flow rate, creek water, granitic pebbles in mesh baskets as substrate | Kruskal-Wallis + non-parametric multiple comparisons | No | Macro-invertebrate community structure | Altered macro-invertebrate community structure (mean abundance) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 9. Laboratory | Species - pulmonate snail, *Planorbella trivolvis* | Exposure (adult mortality / adult growth rate (18d) / gross fecundity / time for embryo development / % embryo abnormality / juvenile survival, pH) | Taunton River water as experimental medium, aquaria, water changed every 11-14 days, photo-period 12h:12h, fluorescent ceiling lights, 19-23°C lab. temp., fed 2x / day on Tetramin, range of $Ca^{2+}$ levels | ANOVA, Fisher's PLSD, Scheffe''s F-test | No | Population | - Mortality i.e. numbers / pH (probability)<br>- Growth rate (max. shell diameter or live weight / pH)<br>- Fecundity (eggs per parent / pH)<br>- Embryo development (time / pH)<br>- % embryo abnor-mality (aberrant gross deformities or abnormal pigment-ation or decomposi-tion or arrested development for >2 wks / pH)<br>- Juvenile survivorship (no. / pH) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 10. Laboratory - multispecies microcosms | Protozoan communities | Exposure (species richness, biomass, ability to sequester nutrients / pH) | Microcosms, colonized poly-urethane foam artificial substrate, dechlorinated tap water as exp. medium, photo-period 16L:8D, full spectrum fluorescent lighting (5000 lux), ambient water temp. (±19.9°C), toxicant replaced every hour, 21 days. | pH response models, LOEL (lowest-observable-effect-level)<br><br>Multiway ANOVA + one-tailed Dunnett's test: species richness.<br><br>Hendrickson's M statistic: taxonomic composition. | No | Protozoan communities | Taxonomic richness (no. of species)<br><br>Taxonomic compo-sition (individuals per species per sample) |
| 11. Laboratory | Species of blue-green algae - *Aphanizomenon gracile* + *Oscillatoria redekei* | Exposure (specific growth rate + copper, pH) | Culture medium, semicontinuous culture (turbi-dostat) to reach continuous exponential growth, 5 $CuCl_2$ concen's per pH value, 8-10 days | $EC_{100}$ graphically estimated, significance of differences between treatments tested according to Nemenyi | No | Organism | Specific growth rate (μ) (biomass / pH value), (biomass / pH reduction and enhanced Cu toxicity) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 12. Field + laboratory | Species - *Amnicola limosa* | Exposure (fecundity, juveniles produced per egg, embryonic development, pH) | Lake buffering capacity, natural populations.<br><br>Static bioassay, complete renewal every 48 hours, 21°C, 16h light:8h dark, defined test solution, *A. limosa* eggs from field | - Log-probit plots for days to hatch or death.<br>- Two-way unbalanced ANOVA. For hatching success, day to hatch + length of hatch.<br>- Multiple linear regressions for variance in fecundity.<br>- PCA for lake water chem-ical variables. | No | Organism | - Fecundity (no. eggs/female in lakes of various pH levels)<br><br>- Number of juveniles produced / egg<br><br>- Embryonic development |
| 13. Laboratory - recirculating artificial streams | Species - caddis fly *Lepidostoma liba*, isopod *Asellus intermedius*, snail *Physella heterostropha* | Exposure (survival + instantaneous growth rates, pH) | Stream water, field organisms in cages in streams, 2°-22°C (seasonal fluctuation), accli-mated for 2 wks, fed on leaf detritus | - Student *t* test for survival.<br><br>Instantaneous growth rate equations. | No | Organism | - Survival i.e. no. at pH 4 vs. control (probability)<br><br>- Instantaneous growth rates (increase in length at pH 4.0 vs. control / time) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 14. Field | Population - *Deleatidium* spp. larvae (Leptophlebiidae) | Population dynamics (incl. density, biomass, annual production), gut contents and feeding rates (pH) | Sampling regime, two naturally acid (pH 4.8) brown-water streams + two alkaline (pH 7.5) clearwater streams | Annual production: size-frequency (Hynes) method.<br><br>Mean annual densities calculated for size classes and converted to biomass with the dry weight to head width relation-ship of Winterbourn (1974). | No | Population | - Mean densities (no. / $m^2$)<br>- Mean total biomass (g larval dry weight (LDW) / $m^2$)<br>- Annual production (g LDW / $m^2$ at various sites)<br>- Gut content (% of food type per total food items)<br>- Grazing rates on epilithon (amount + composition (organic/inorganic) material) |
| 15. Laboratory | Species - the midge *Tanytarsus dissimilis* | Exposure (life cycle, pH) | Static bioassays in glass aquaria, lake water, room temp. ± 18.5°C, fed by a mixed algal culture, 16h light : 8 h dark cycle, 35 days, range of pH values | | No | Organism | Mortality (no. dead / pH). (* The life cycle could not be completed below a pH of 5.5) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 16. Laboratory | Natural populations - *Daphnia pulex*, *D. galeata mendotae*, *Simocephalus serrulatus*, *Mesocyclops edax*, *Chaoborus americanus*, *C. punctipennis* | Exposure (pH (4.0 - 6.0), time) | Sampling regime, 48h or 96h acute bioassays in aquaria, pond water, incubation at ±0.5°C of collection temp., exp. temp. from 15-22°C, 16h light: 8h dark, no feeding or aeration | Probit analysis + trimmed Spearman-Karber for determining LC50 values | No | Population | Mortality i.e. % over time (probability) |
| 17. Field | Community - benthic algae | - Taxonomic unit (correlation with nutrient enrichment at two sites of differing pH (gradient of alkaline to acid).)<br><br>- Exposure (growth rate + average cell size, pH + increased nutrient levels) | Sampling regime, nutrient addition by adding substrata (releasing either N or P), 21 days, single habitat, 2 pH sites | Two-way ANOVA, Student Newman-Keuls range test, correlations using SAS-82 | No | Benthic algal community structure + population (algal growth) | Benthic algal growth + altered community structure ( total biovolume + chlorophyll-*a* accumulation) |
| 18. Laboratory | Species - brown trout *Salmo trutta* | Exposure (accumulation of $^{54}$Mn, pH) | Glass aquaria, artificial aerated freshwater, water changed every 3-4 days, pHs of 5.3 + 7.5, 0.1 μg/L Mn(II) (as $MnCl_2$) with 1.5 μCi/L $^{54}MnCl_2$ | *T*-test from the SAS/STAT software | No | Organism | $^{54}$Mn uptake ($^{54}$Mn concen. in organs + tissues / time) |

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 19. Field | Population - crustacean zooplankton species | Exposure (nutrient enrichment, pHs of 11.2, 10.8 and 10.4) | Natural ponds, nutrient addition i.e. agricultural fertilizer, triple superphosphate, ammonium nitrate, muriate of potash at a ratio of 8:2:1 (N:P:K) | Size-frequency distributions used to estimate the population dry weight biomass | No | Population | - Zooplankton mortality (probability) |
| 20. Laboratory + field | Amphibian populations | Exposure (pH, time) | Short-term effects + field distributions | | No | Populations | - Mortality (probability)<br>- Hatching success<br>- Egg fertilization<br>- Egg production<br>- Egg development<br>- Embryo development<br>- Distribution |

## References


1. Riedel G F, and Sanders J G (1996) The influence of pH and media composition on the uptake of inorganic selenium by *Chlamydomonas reinhardtii*. Environmental Toxicology and Chemistry **15 (9)**: 1577-1583.


**Results**

Little difference was noted in the uptake of selenate as a function of pH, with the maximum uptake occurring at pH 8. The variations across pH appeared not to be systematic.

Selenite uptake increased substantially at the lower pH values.

**Background literature**

Adverse environmental effects appear to result largely from transfer of selenium from lower to higher trophic levels, especially via uptake by phytoplankton.

**Predictability**
Unknown
**Limitations**
Authors suggest results can be extrapolated to phytoplankton and the subsequent transfer of selenium to higher trophic levels.

2+3. Rousch J M, Simmons T W, Kerans B L, and Smith B P (1997) Relative acute effects of low pH and high iron on the hatching and survival of the water mite (*Arrenurus manubriator*) and the aquatic insect (*Chironomus riparius*). Environmental Toxicology and Chemistry **16 (10)**: 2145-2150.

**Results**
Egg hatching was reduced at pH 2 for midges and at pH 3 for mites. Survival of midge larvae was partially reduced at pH 4, and survival of mite deutonymphns, larvae, female and male adults was reduced at pH 3.
*A. manubriator* is an ideal water mite species to use in toxicity tests as it is easy to maintain in the laboratory and has readily distinguishable life stages. However, these acute exposure experiments will not reveal sub-lethal effects that may have confounded results.
**Background literature**
Field studies show that some water mites are sensitive to low pH, to agricultural pollution and domestic-industrial toxins, and to acid mine drainage. It has been proposed (Havas and Hutchinson, 1983) that the ability of aquatic organisms to survive polluted conditions, specifically low pH, may be related to the successful osmoregulation of ions.
**Predictability**
Unknown
**Limitations**
Results limited to acute toxicity; there is no measure of sub-lethal effects that may have confounded results.
Supporting literature suggests results may be extrapolated to include other water mites, but does not specify which species.

4. Rutgers M, van Bommel S, Breure A M, van Andel J G, and Duetz W A (1998) Effect of pH on the toxicity and biodegradation of pentachlorophenol by *Sphingomonas* sp. strain P5 in nutristat culture. Environmental Toxicology and Chemistry **17 (5)**: 792-797.

**Results**
During steady state conditions, PCP exerted a stronger toxicity on the growth of *Sphingomonas* sp. strain P5 at lower medium pH levels than at higher pH levels. Inhibition of growth of strain P5 by PCP was correlated to the concentration of undissociated phenol in the system, rather than the total PCP concentration. Strain P5 is essentially a neutralophilic bacterium.
The equations describing growth kinetics, biodegradation kinetics and pH optimum curves were fitted to the experimental data using nonlinear regression analysis based on the algorithm of Marquardt.

**Background literature**
Few literature reports dealing with the effect of pH on toxicity and degradation of chlorinated phenols by microorganisms exist. Available reports cannot be easily compared to this study as most studies were conducted as acute exposures to PCP, and not at steady-state.
**Predictability**
Unknown
**Limitations**
The study is specific to *Sphingomonas* sp. strain P5 at steady-state conditions. However, the sensitivity for chlorophenols was found not to be significantly different between chlorophenol-degrading and chlorophenol-nondegrading strains. The authors do not expect the effect of pH on the toxicity of chlorophenols to be unique as the target of toxic action is the phospholipid membrane.

5. Mackay R J, and Kersey K E (1985) A preliminary study of aquatic insect communities and leaf decomposition in acid streams near Dorset, Ontario. Hydrobiologia **122**: 3-11.

**Results**
Eight streams showing similarity in size, substrate and current velocity were selected from a study of 19. Invertebrates were identified to genus, some only to family level. Benthic invertebrates were dominated by insects in terms of abundance and taxa. Generic diversity and abundance showed trends which paralleled the gradient in pH i.e. the highest number of genera in the least acid stream. Ephemeroptera were most susceptible to increasing acidity, and Plecoptera also in low numbers. Caddisflies were more tolerant, with Trichoptera being the most tolerant of increasing acidity. Shredders were the most common macroinvertebrates.
**Background literature**
A study in Sweden showed that shredders were the commonest macroinvertebrates in an acid stream. This ability may have evolved as an adaptation allowing shredders to cope with locally acid conditions in patches of decaying organic matter. The adaptation then allowed them to colonize acidic habitats on a large scale.
**Predictability**
Unknown
**Limitations**
The relationship could be extrapolated to other field sites, but will require testing. This paper also refers to northern hemisphere streams, so the relationship will have to be tested for South African invertebrates.

6. Wade K R, Ormerod S J, and Gee A S (1989) Classification and ordination of macroinvertebrate assemblages to predict stream acidity in upland Wales. Hydrobiologia **171**: 59-78.

**Results**

A range of environmental data (including substratum type) were collected from riffles at 104 sites. Taxa were identified to species where possible. Relationships between TWINSPAN classification and environmental variables were undertaken by MDA. The number of taxa collected varied greatly between sites, and correlated most strongly with pH and aluminium concentration. There was some indication of a relationship between taxon richness and land-use.

**Background literature**

Many studies have showed impoverished macroinvertebrate fauna in base-poor, acidic streams. Studies have also related certain invertebrate groups to stream acidity and seasonal variations in pH. The results of this study were consistent. However, this data does not determine the nature of any causal relationships between pH-related factors and macroinvertebrates.

**Predictability**

Yes. As all environmental data is used, any uncertainties in the output could be ascribed to uncertainties in the input.

**Limitations**

This procedure / method could be followed elsewhere, and the relationships tested for South African conditions.

7. Zischke J A, Arthur J W, Nordlie K J, Hermanutz R O, Standen D A, and Henry T P (1983) Acidification effects on macroinvertebrates and fathead minnows (*Pimephales promelas*) in outdoor experimental channels. Water Research **17**: 47-63.

**Results**

Acidification did not markedly increase toxic metal concentrations in the channels. Benthic macroinvertebrate densities were lower in the acidified channels - final diversity indices were 2.1, 1.7 and 1.2 for pH 8 (ambient), pH 6 and pH 5 respectively, however, changes in community structure occurred in all channels during the study. Macroinvertebrate tolerances to acidification were classified as follows: damselflies, isopods and leeches most tolerant, chironomids; some amphipods and flatworms of some tolerance; and other amphipods (e.g. *Hyalella azteca)* and snails most sensitive. Acidification stimulated macroinvertebrate drift activity. Under conditions of acid stress, emergence of aquatic insects appears to be the critical stage of the life cycle. Fathead minnow spawning and embryo production similar in pH 8 (ambient) and pH 6 channels; no spawning or eyed-embryos in the pH 5 channel. Newly-hatched juvenile fish did not survive to the juvenile stage in the pH 6 channel. Results therefore suggest that no-effect pH values would be greater than 6.0 in the outdoor channels.

The non-parametric Kruskal-Wallis test was used to compare stations within the three channels. ANOVA was the parametric test for inter-channel comparisons. Diversity indices were calculated.

**Background literature**

Several groups have recommended pH levels required to protect aquatic life. A recommended pH level above pH 6.5 was recommended by the National Academy of Sciences and the US EPA for the maximal protection of aquatic life. Few studies have been conducted on the acidification effects in natural streams when not complicated with heavy metal additions from acid mine wastes. Generally studies show that increased pH levels show a reduction in diversity and density of the benthos, decreased emergence and increased drift.

**Predictability**

Unknown

**Limitations**

Data were developed using Mississippi River water as experimental medium and support the pH recommendations of the National Academy of Sciences (1973) and the US EPA (1976). As *Pimephales promelas* is a northern hemisphere fish species, results would have to be tested for South African conditions

8. Allard M, and Moreau G (1987) Effects of experimental acidification of a lotic macroinvertebrate community. Hydrobiologia **144**: 37-49.

**Results**

This study encompassed the acidification of communities, with and without aluminium. Experimental conditions were designed to discriminate between the effects produced by a reduced pH, and those produced by the liberation of metals. Under acidification (pH 4.0 vs. reference pH of pH 6.3 - 6.9) the mean abundance of all groups of organisms were lowered. Mayflies nearly completed disappeared from the channels. Authors suggest the results are due to the direct action of hydrogen ions through physiological stress. Early instars were particularly sensitive to low pH. The only organism not affected was *Microtendipes* sp.. Organisms buried inside the artificial substrate had a delayed reaction to acidification. Resistance of invertebrates vary within taxonomic group and life stage.

Analysis was by Kruskal-Wallis statistical tests, followed by non-parametric multiple comparisons.

**Background literature**

Numerous studies have been performed to assess the effects of acidic precipitation on benthic invertebrates in North America and Scandinavia. These include 1) surveys of lakes affected by acid precipitation, 2) comparisons between acidified and non-acidified lakes, 3) experimental acidification of rivers, and 4) experimental acidification in artificial streams. Unfortunately studies have often produced conflicting results, due to factors such as possible synergistic metal effects, the length and strength of acid pulses, and consideration of the state before acidification.

**Predictability**

Unknown

**Limitations**

Results are limited to creek water (oligotrophic and poorly buffered) due to possible speciation effects, and would have to be tested in other systems.

9. Hunter, R D (1990) Effects of low pH and low calcium concentration on the pulmonate snail *Planorbella trivolvis*: a laboratory study. Canadian Journal of Zoology **68**:1578-1583.

**Results**

Adult snail mortality was negligible. Both low pH treatments (pH4.6 + pH4.7) resulted in significantly reduced snail growth rates and gross fecundity.

None of the eggs laid in low pH treatments hatched, mostly due to developmental arrest. Embryos and juveniles therefore showed much greater sensitivity than adults.

**Background literature**

Molluscs are among the groups of freshwater animals most sensitive to acidification. Reviews indicate that both gastropods and bivalves are relatively intolerant to low pH. Previous studies suggest that disappearance due to acidification is related to recruitment failure. This could be due to low $Ca^{2+}$ levels present in acidified systems.

**Predictability**

Probably, as there appears to be a direct relationship between low pH and effect. However, this study did not cover a range of pHs, but rather two selected low pHs (pH 4.6 + 4.7) vs. two neutral treatments (pHs 7.2 + 7.4).

**Limitations**

This result is only reported for this single species, but authors suggest a similar effect on other pulmonate species.

10. Niederlehner B, and Cairns J Jr (1990) Effects of increasing acidity on aquatic protozoan communities. Water, Air and Soil Pollution **52**: 183-196.

**Results**

Taxonomic composition of protozoan communities was significantly affected even with slight changes in pH. The group exposed to a pH of 3.33 had only one species in common with the control group. There was no significant differences from the control in biomass. The mean value in the pH 3.3 group was much lower, but not significant. The IC20 for species richness from this study predicted that 20% of taxa would be adversely affected at a pH of 6.92.

**Background literature**

Many surveys of natural systems with different pHs have shown a successive loss of species richness with increasing acidity for fish, zooplankton, phytoplankton and benthic algae. Schindler (1987) found that the community composition of small, rapidly reproducing species with wide dispersal powers e.g. phytoplankton, were among the most sensitive indicators of acidification, while primary production, decomposition and nutrient processing remained unchanged to pH 5. Field studies have suggested that the most serious decreases in taxonomic richness occur at pH values < 5.6.

**Predictability**

Unknown

**Limitations**

Results would have to be tested for South African conditions.

11. Lüderitz V, and Nicklisch A (1989) The effect of pH on copper toxicity to blue-green algae. Int. Revue. Ges. Hydrobiol. **74**: 283-291.

**Results**

*Aphanizomenon* showed a greater growth rate decline with pH lowering than *Oscillatoria*. Lowering of pH leads sooner to copper toxicity enhancement in *Oscillatoria* (at pH 7.2) than *Aphanizomenon* (at pH 6.2), however, a combination of copper addition and pH shock does not lead to a lowering of effective copper concentration. At pH 5.1, shortening of the interval between copper toxicity and copper stimulus is characteristic of both species. It seems possible that $H^+$ ions play a protective role against copper ions and that toxicity starts only at a distinct intracellular concentrations.

**Background literature**

Metal availability and toxicity to aquatic biota are influenced by pH dependent factors such as binding to organics, precipitation and ionic interactions.

**Predictability**

Unknown

**Limitations**

This result is only reported for these two species, but authors report similar results for other algal species.

12. Shaw M A, and Mackie G L (1990) Effects of calcium and pH on the reproductive success of *Amnicola limosa* (Gastropoda). Canadian Journal of Fisheries and Aquatic Science **47**: 1694-1699.

**Results**

Lake pH (buffering capacity) alone explained 63% of the variation in fecundity in clear lakes. Fecundity vs. pH showed that snails from the lowest pH lake showed unusually high fecundities. While fecundity decreases with pH, juvenile success is impaired only when pH drops below 5.80. pH was not a good predictor of juvenile success in lakes supporting viable populations of *A. limosa*. There is a linear trend between pH and hatching success. There was no significant difference in the average length of newly hatched snails incubated at different pH levels. Laboratory experiments show that snail growth is inhibited at low pH.

**Background literature**

Literature reports that gastropods are very sensitive to acidification. *In situ* experiments show that productivity, growth and survival decline below pH 6.0.

**Predictability**

Unknown

**Limitations**

Results are reported only for *A. limosa* under soft-water conditions. Differences have also been noted in the literature depending on the nature of the test water.

13. Allan J W, and Burton T M (1986) Size-dependent sensitivity of three species of stream invertebrates to pH depression. *Impact of acid rain and deposition on aquatic biological systems, ASTM STP 928*, BG Isom, SD Dennis, and JM Bates, Eds., American Society for Testing and Materials, Philadelphia, pp. 54-66.

**Results**

The vulnerability of the caddis fly *Lepidostoma liba*, the isopod *Asellus intermedius*, and the snail *Physella heterostropha*, to a change in pH from pH 6.7-

7.2, to pH 4.0, was directly correlated to size, with smaller individuals being much less tolerant to pH 4.0 depression than larger individuals. For all size classes of *Asellus* and *Lepidostoma*, mortality was increased at pH 4.0, but rate of mortality was size dependent. Growth rate of all *Asellus* size classes was slower at pH 4.0, but only slightly reduced for *Lepidostoma*. Mortality of *Physella* was correlated with shell size. Mature specimens (12mm shell size) survived equally in pH 4.0 and control streams.

**Background literature**

Acidification of aquatic ecosystems to pH 5 or less can cause mortality for many species of invertebrates. Other effects include decreased emergence, increased drift, increased food consumption and decreased growth. Effects appear to vary with size and maturity.

**Predictability**

Unknown

**Limitations**

The necessity for testing a range of life stages with a special emphasis on smaller size classes is probably valid for most organisms.

14. Collier K J, and Winterbourn M J (1990) Population dynamics and feeding of mayfly larvae in some acid and alkaline New Zealand streams. Freshwater Biology **23**: 181-189.

**Results**

Mean densities of larvae were higher in alkaline streams (pH 7.5). Mean biomass was always highest at the stable, spring-fed, alkaline site and was lower at the acid sites (pH 4.8) and another alkaline site where the population was always dominated by small larvae. Annual production was high at the more stable, alkaline site. Gut contents were dominated by fine particulate matter (69-99%), diatoms (up to 21%), and filamentous algae (8-13%). Grazing rates on epilithon were higher on stones taken from acid than alkaline streams, and material grazed from acidic systems contained a higher proportion of inorganic material. Higher grazing rates may reflect lower quality of epilithic food in acid streams, resulting in lower *Deleatidium* populations.

**Background literature**

Mayflies are generally absent or poorly represented in acidic streams in the Northern hemisphere. It was shown that experimental acidification resulted in decreased growth and population size of *Ephemerella funeralis*, but also that the absence of *Ephemerella ignita* was mostly due to the decreased diversity of food resources.

**Predictability**

Unknown

**Limitations**

The relationship is expressed specifically for *Deleatidium* populations in some acid and alkaline New Zealand streams only.

15. Bell H L (1970) Effects of pH on the life cycle of the midge *Tanytarsus dissimilis*. The Canadian Entomologist **102**: 636-639.

**Results**

At the end of 5 days, all larvae were dead at pH 3.0. Several at pH 4.0 were dead and the rest were under stress. At the end of 10 days, all larvae at pH 4.0 were dead. At pH 5.0 all larvae survived and began pupating after 13 to 17 days. At pH 5.5, 6.0 and the control (pH 7.8), all larvae survived and began pupating in 12-14 days. After 27 days, adults began to appear at pH 5.5, 6.0 and the control. All 15 larvae from each concentration emerged successfully within 35 days. At pH 5.0, no adults had emerged within 35 days. All tests at pH 5.5, 6.0 and the control were continued, and three complete generations were obtained.

**Background literature**

Literature states that aquatic insects are generally tolerant of acid conditions, at least for periods of less that 1 week's duration. Values as low as 1.5 are tolerated by certain caddisflies for 96 hours. Long-term studies (30 days) have given values as low as pH 2.45 for certain aquatic insects.

**Predictability**

Unknown

**Limitations**

Results are expressed for this species only, although authors state that test organisms died at pH values found in acid mine streams, some industrial wastes, and peat bogs.

16. Price E E, and Swift M C (1985) Inter- and intra-specific variability in the response of zooplankton to acid stress. Canadian Journal of Fisheries and Aquatic Science **42**: 1749-1754.

**Results**

Cladocerans were most susceptible to acid stress, followed by *Mesocyclops* and *Chaoborus* larvae. *Simocephalus* was the most tolerant cladoceran, follwed by *D. pulex*, then *D. galeata mendotae*. *Daphnia* populations tested in the spring or early summer were more tolerant of low pH than those tested in the fall. *Mesocyclops edax* from an acid pond were more tolerant than those from a neutral pond. The 96h LC50 for the *Chaoborus* species (2.00, 2.09) were two pH units lower than those of the cladocerans, and one pH unit lower than that of *M. edax*.

**Background literature**

Studies have shown that acidification simplifies zooplankton communities. A sharp decrease in the abundance and diversity of zooplankton is evident below pH 5.0. **Predictability**

Unknown

**Limitations**

Although acute studies on other species are needed to verify the range of variability in zooplankton response to acidification, these results suggest interspecific variability in acid sensitivity and suugest intraspecific variability due to habitat and season. Life history stage was not addressed, and chronic studies of the trophic structure of lakes under acid stress are also required.

17. Carrick H J, and Lowe R L (1989) Benthic algal response to N and P enrichment along a pH gradient. Hydrobiologia **179**: 119-127.

**Results**

Progression toward a community composed of larger cells following P enrichment observed along the pH gradient, seems to be related to the dominance of larger celled filamentous green algae Benthic algal growth therefore seems to be strongly linked to enrichment with P. Although chlorophyll-*a* concentrations were higher under more acidic conditions, nutrients exhibited greater control on benthic algal growth than change in hydrogen ion concentration (pH).

**Background literature**

In addition to nutritional factors, physico-chemical parameters such as hydrogen ion concentration are strong selective factors in aquatic habitats. The response of benthic algae to nutrient perturbation is not as well understood as phytoplankton.

**Predictability**

Unknown

**Limitations**

Results are site-specific, but principles are of general application.

18. Rouleau C, Tjälve H, and Gottofrey J (1996) Effects of low pH on the uptake and distribution of $^{54}$Mn(II) in brown trout (*Salmo trutta*). Environmental Toxicology and Chemistry **15 (5)**: 708-710.

**Results**

At the end of the experiment, the whole-body $^{54}$Mn of fish held in water at pH 5.3 was significantly higher (1.7 times) than fish held at pH 7.5. A similar increase was observed in viscera, brain and eyes. These results suggest that higher Mn levels found in fish living in acidic waters may not only depend on the higher total Mn concentrations usually found in these waters. Results show that fish could accumulate an additional 1.3-4.6 mg Mn/kg at low pH.

**Background literature**

It is known that acidification increases Mn concentrations in water, which has been associated with a parallel increase of Mn concentration in fish. However, increased fish tissue Mn concentrations have also been noted for fish in circumneutral waters. It is therefore not clear whether increased uptake results solely from higher Mn concentration or also from a pH-related modification of some processes associated with the uptake or bioavailability of Mn.

**Predictability**

Unknown

**Limitations**

Results expressed are specific to this fish species.

19. O'Brein W J, and DeNoyelles F Jr (1972) Photosynthetically elevated pH as a factor in zooplankton mortality in nutrient enriched ponds. Ecology **53 (4)**: 605-614.

**Results**
Results show close agreement between high pH values and the disappearance of crustacean zooplankton in some of the highly-fertilized ponds. High pH levels were stimulated by high levels of primary productivity. A change from pH 10.8 to 11.2 radically altered survivorship of zooplankton. It is suggested that in culturally enriched waters of low buffering capacity such a pH mortality factor may be common.
**Background literature**
It has been proposed that pH levels of natural ponds may be important in determining the abundance of *Daphnia*. It has also been shown that copepods and cladocerans are little influenced by the pH of water, although cladocerans have a maximal feeding rate at certain optimal pH levels.
**Predictability**
Unknown
**Limitations**
Loss of zooplankton from lakes may have dramatic results as they form the major pathway through which large amounts of phytoplankton may be channeled to other trophic levels. The disappearance of zooplankton, and associated increase of phytoplankton, is a fact which is probably valid for many systems.

20. Herfenist A, Power T, Clark K L, and Peakall D B (1989) A review and evaluation of the amphibian toxicological literature. Technical Report Series No. 61, Canadian Wildlife Service, pp. 222.

**Review of literature**
This review only addresses the direct effects of toxicity of acidity. Short-term effects of elevated hydrogen ions have been observed at all life stages. Fertilization of *Rana pipiens* eggs was reduced below pH 6.5. Below pH 4.8 there was complete failure of egg development. The optimal pH for normal egg development was above pH 6.0. Levels of pH that would allow 50% or more embryos to survive in laboratory bioassays for 11 species ranged from 4.7-3.8. Short-term pH depressions were also toxic to developing embryos. Optimal development of amphibians is dependent upon both pH and temperature. pH affects the distribution of many amphibian species and was found to be one of the most important habitat characteristics. Most species avoid acid waters. Affects may be due to disruptions in ionic balance.
**Predictability + Limitations**
Unknown

**ELECTRICAL CONDUCTIVITY (EC) / SALINITY / TOTAL DISSOLVED SOLIDS (TDS)** *(SCHERMAN)*

| INPUT | | | RELATIONSHIP | | | OUTPUT | |
|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncertainty calculation | Level | Type (Metric) |
| 1. **Salinity** - field | Benthic community | Taxonomic unit (presence/absence) | Sampling regime | Rule base | No | Aquatic ecosystem | Ecosystem integrity under saline conditions |
| 2. **Salinity** - field | Communities + populations | Exposure (salinity level) | Lethal + sub-lethal effects: Exposure conditions | Rule base | | Taxonomic unit | Survival (probability) |
| 3. **EC** - laboratory | Species - mayfly *Tricorythus* sp. + caddisfly *Chimarra* sp. | Exposure (EC level, time) | Recirculating artificial streams, 96h acute tests, natural populations, river water, elevated EC by NaCl + $Na_2SO_4$ daily 20% renewal of test medium, lab. temp. 14-23°C, 12h:12d. | LC50s determined by probit analysis | No | Organism | Mortality (probability) |

**REFERENCES**

1. Hart B T, Bailey P, Edwards R, Hortle K, James K, McMahon A, Meredith C, and Swadling K (1996) Effects of salinity on river, stream and wetland ecosystems in Victoria, Australia. Water Research **9**: 1103-1117.

**Results**

Macroinvertebrates and plants (riparian vegetation, macrophytes and microalgae) were assessed to be the most salt sensitive biological communities, with direct adverse effects likely to occur when salinity is increased to around 1000 mg/l. More subtle sub-lethal effects may occur at salinities below this value.

**Background literature**

In Australia, the following relationship is used to relate between total soluble salts (mg/l) and EC (µS/cm):

Total soluble salts = 0.68 x EC

**Predictability**

Unknown

**Limitations**

Information presented is limited to Australian river, stream and wetland ecosystems, but may be valid for South African conditions.

2. Hart B T, Bailey P, Edwards R, Hortle K, James K, McMahon A, Meredith C, and Swadling K (1991) A review of the salt sensitivity of the Australian freshwater biota. Hydrobiologia **210**: 105-144.

**Review of literature**

The review highlights a general lack of data on the sensitivity of freshwater plants and animals to salinity increases, but shows that direct adverse effects are likely to occur if salinity levels are increased to around 1000 mg/l. Microbial community: small changes in salinity have little effect on microbial processes, unless combined physical -saline (e.g. stratification) events occur. Macrophytes: Many are salt sensitive, with 1000 - 2000 mg/l being lethal. Many sub-lethal responses are seen. Riparian plants: Adverse effects are often apparent above 2000 mg/l. Invertebrates: Invertebrates are most sensitive to increasing salinities, with adverse effects apparent at 1000 mg/l. Most sensitive insects include stones, some mayflies, caddisflies, dragonflies and water-bugs. Most sensitive molluscs are pulmonate gastropods. Fish: Fish are generally tolerant to salinities >10 000 mg/l. Larval fish are more sensitive than adults, and eggs more tolerant than larvae. Tadpoles + amphibian egg-masses: Sensitive indicators in wetlands. Freshwater turtles most at risk. Waterbirds: Evidence of low breeding success where salinity > 3000 mg/l.

**Predictability + Limitations**

Unknown

3. Goetsch P-A, and Palmer C G (1997) Salinity tolerances of selected macroinvertebrates of the Sabie River, Kruger National Park, South Africa. Archives of Environmental Contamination and Toxicology **32 (1)**: 32-41.

**Results**

Results indicated that mortality could not only be linked to increasing EC/TDS levels, but also to the nature of the salt used to elevate EC. Sodium sulphate (LC50=100 mS/m) was shown to be more toxic to the mayfly *Tricorythus* sp. than sodium chloride (LC50=400-800 mS/m).

**Predictability**

Unknown

**Limitations**

Results are species and site-specific. No attempt has been made to extrapolate to other organisms or rivers. The applicability of the results to other systems would have to be tested.

**DISSOLVED OXYGEN** (*MULLER*)

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| Laboratory | Rainbowfish embryos | Low oxygen (2-3 mg/l) for 6 days, in conjunction with various levels of exposure to endosulphan | 25°C<br>pH 6.8-6.9<br>EC 80-90 $\mu S$ | | | organism | embrionic development: birth defects and larval mortality; longer developmental period | Barry *et al.* 1995 |
| Field study, using tagged fish | chinook salmon | supersaturation (dams); > 110% causes gas bubble disease | | | no | organism | behaviour: avoided the supersaturated water | Gray 1990 |
| field and lab studies | *Pimephales promelas*: caged in streams, downstream of impact sites | 24 hour exposure; (48 hr acute static tests as well); low oxygen levels (20% saturation/ 1.7mg/l) adversely affected ability to survive: may have increased susceptibility to chlorine. | 8-14 days old; acclimated to stream conditions for 1-2 days; water velocity <0.27m/s; | | | organism | survival | Szal *et al.* 1991 |
| laboratory | *Asellus aquaticus* and *Gammarus pulex* (adult males and juveniles) | 24 hr exposure; *A. aquaticus*: 0.32 and 0.25mg$O_2$/ℓ; *G. pulex*: 1.63 and 1.26mg$O_2$/ℓ. | field collected: acclimated for at least 1 week; 15°C | 24hr LC50, least square regression technique | | species and organism: males and juveniles; *Asellus* more sensitive than *Gammarus*, adults more sensitive than juveniles | mortality, ventilation rates (beats per minute); also measured respiratory surfaces and haemolymph | Maltby 1995 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc | Level | Type | |
| (unknown) | *Daphnia magna* and *D. pulex* | can survive $O_2$ levels as low as 0.1mg/ℓ (estimated LC50); can thrive at levels above 0.5mg/ℓ | ? | estimated 24 hr values, on a sigmoidal curve; model for a wastewater pond, which assumes various ecological parameters, which are not true. | ? | | | Hathaway and Stefan 1995 |
| laboratory | *Cyprinus carpio* (common carp) | at the critical oxygen level of 45μmol/ℓ (1.44mg/ℓ) the $O_2$ consumption decreased and the fish were unable to regulate their oxygen consumption. | aquaculture fish, reared to 15-30g; pH 7-7.8; fish starved for 2 days prior to testing; overnight acclimation to the test chamber; standard water; | ANOVA | | organism | physiological stress: oxygen consumption rate decreases as the fish are no longer able to regulate their oxygen intake | De Boeck *et al.* 1995 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| laboratory study | Stoneflies: *Pteronarcys dorsata*, *Acroneuria lycorias* Mayflies: *Hexagenia limbata*, *Ephemera simulans*, *Ephemerella subvaria*, *Baetisca laurentina* Caddisfly: *Hydropsyche betteni* Midge: *Tanytarsus dissimilis* | Range of LC50 values, ranging from <0.6mg/ℓ to 3.9mg/ℓ. Tolerance was affected by exposure time: LC50's increased with longer exposure time: 4.5-5mg/ℓ (the midge tolerance level remained at <0.6mg/ℓ). Require a minimum of 6mg/ℓ for 50% emergence success (depending on species). | Used lake water as the test medium; pH 7.5-7.8; most tests carried out at 18.5°C. | LC50, with confidence limits | | differences in responses between species. | survival: differences between species; in 1 of the species (only 1 tested), the tolerance was affected by temperature: lower water temperature meant the animals were able to tolerate lower oxygen levels. DO levels affected adult emergence rates: lower DO levels resulted in lowered emergence rates. | Nebeker 1972 |

**TEMPERATURE** (*MULLER*)

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| Field study, using tagged fish | chinook salmon | temperature (heated effluent) | increased temperatures enhanced negative effects of other chemicals | | | organism | behaviour: avoided temperature differences > 9 - 11°C | Gray 1990 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| laboratory study | 2 species mayfly (*Ephemerall invaria*, *Stenonema ithaca*); 2 species caddisfly (*Symphitopsyche morosa*, *Brachycentrus lateralis*): all collected from riffle areas | collected from stream, allowed to acclimate for 3 days in artifical stream, exposed to heated test streams | 96hr exposure at increased temperature; | LT50 and regression coefficient | | species | differences in thermal tolerances between all the species, ranging from 22 - 32°C, depending on species | deKozlowski and Bunting 1981 |
| laboratory study | 3 species of phytoplankton: *Chlorella vulgaris* *Synechococcus* sp *Isochrysis galbana* | taken from lab cultures | 4 hrs incubation; temp range: 5, 10, 15, 20, 25, 30, 35, 40, 45 under constant light. | | | species | % DOC excretion: a measure of photoinhibition: differences in responses by the different species: inhibition of growth >30°C in all species, and <15°C in 1 species | Zlotnik and Dubinsky 1989 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| laboratory study | *Pimephales promelas* | tests began with 2 day old fry, laboratory reared (apparently acclimated to 21°C). | Experimental water was a mixture of pond water and carbon-filtered, demineralized tap water; 16-hr photoperiod; test temperatures were: control (fluctuating), 34, 32,30, 28, 26°C, with associated differences in DO. > 28°C resulted in adverse effects (death, no spawning/hatching); <22°C resulted in reduced spawning. | ANOVA | | organism | Almost 100% mortality at 34°C. Reduced growth and gonad development at 32°C, although increased survival. Reduced secondary sexual characteristics, reduced spawning, no hatching at 30°C. No noticeable adverse effects at 26, 28 and control temperatures, although reduced spawning below 22°C; spawning and egg hatching took place. | Brungs 1971 |
| laboratory study | *Isonychia* nr. *sadleri* (Ephemeroptera); *Hydropsyche* spp. (Trichoptera) | increasing temperatures, from different acclimation temperature starting points, for different exposure times. | Field collected animals, acclimated for 5-7 days. Used dechlorinated tap water; held in aerated containers before use. | Two-sided Student's *t*-test | | species | mortality, moulting. Differences in thermal tolerances between the species tested: *Hydropsyche*: no mortality <28°C, significant mortality 36-38°C. *Isonychia*: no mortality <26.5°C, significant mortality 33.5-35°C. | Sherberger *et al.* 1977 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| laboratory study | *Goniobasis livescens Lymnaea emarginata* | Exposed to increased temperature after exposure to sublethal levels of Zn and Cr: measured time it took for the animals to die. Animals were exposed to increased temperature for up to 1 hr, after being exposed to toxicant for 58-60 hrs. | Field collected animals; experimental medium was lake water. Tests and holding water was 24 °C. | ANOVA | | species | mortality. Exposure to sublethal levels of toxicants increased the mortality of *G. livescens*, i.e. they died faster. 38° resulted in increased mortality rate when compared to 35°. *L. emarginata* were more tolerant to higher temperatures than *G. livesencs*. | Cairns and Messenger 1974 |
| laboratory study | Used 5 clones of *Daphnia magna* | Exposed the cultures to 14, 19, 24 and 29°C; measured number of living and dead animals, the sex of the offspring, size at death, average lifespan. These were used to generate life table statistics, generation time, intrinsic rate of increase (according to formulae given). | Initally field collected females, but maintained 5 clone cultures separately in the laboratory for testing. Maintained cultures at 22°C, in aged tap water. | ANOVA for the growth, survivorship and reproductive values: Kruskal-Wallis test. (should have used range-finding tests as well). | | clones/ organism | Life history parameters: growth, survivorship and reproduction. Significant differences between clones. | Korpelainen 1986 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type | Parameter | Form | Uncert. calc. | Level | Type | |
| laboratory study | tested 7 species, taken from ponds and streams: *Neurocordulia alabamensis* (CS), *Macromia illinoiensis* (CS), *Epitheca cynosura* (CP), *Ladona deplanata* (CP), *Celithemis* sp. (CP), *Pachydiplax longipennis* (SP), *Libellula auripennis* (SP) (C=control; S=stream; P=pond) | Acclimated at various increased temperature, and obtained critical thermal maximum and lethal temperatures for each species (as a measure of thermal tolerance). Temperature range: 12, 16, 20, 24, 28 and 32°C. Body lengths were measured. | Field collected, from ponds and streams both affected and unaffected (control) by thermal pollution. Stream water was used as the test medium. Acclimated for at least 48 hrs. | ANOVA: species, acclimation temp. and their interaction. Regression analysis between lethal temperature and critical thermal maximum. | | species | The acclimation temperature affected the thermal tolerance: higher acclimation temp. resulted in a higher thermal tolerance. Differences in thermal tolerances between species (weak correlation with body length), depending on where they came from: the animals from the control streams and ponds had the lowest thermal tolerances (38-39°C), while those from the thermal pond had the highest (43-44°C). | Garten and Gentry 1976 |

## REFERENCES


Barry MJ, Logan DC, Ahokas JT and Holdway DA (1995) Sublethal effects of endosulfan and oxygen concentration of embryos of the Australian crimson-spotted rainbowfish (*Melanotaenia fluviatilis*). Australian Journal of Ecotoxicology 1: 71-76.

Brungs WA (1971) Chronic effects of constant elevated temperature on the fathead minnow (*Pimephales promelas* Rafinesque). Transactions of the American Fisheries Society 4: 659-664. Upper lethal temperatures are dependent on the temperatures to which the animals have been acclimated. 2-Day old fry (from water temperatures of approximately 21°C) were introduced into test temperature medium, allowed to grow, spawn and these eggs were maintained at the same test temperatures, allowed to hatch and grow for 90 days. Details of the test chambers, method of water exchange, details of water chemistry, feeding frequencies, measuring, experimental design (pseudoreplication) are given in the paper. Length measurements are not considered a particularly good indicator of the effects of temperature, since there was great variation, and no opportunity in the experimental design to measure the growth rates of individual fish. Some of the effects of the high temperatures may be associated with reduced DO levels associated with increased water temperature.

Cairns J Jr and Messenger D (1974) An interim report on the effects of prior exposure to sublethal concentrations of toxicants upon the tolerance of snails to thermal shock. Archives of Hydrobiology 74 (4): 441-447. Exposed 2 species of snail to sublethal levels (0.2 of the 48-hr LC50 value) of toxicants (Zn and Cr) for 58-60hrs and then tested their reponse to increased temperatures (35 and 38°C). Previous exposure to sublethal toxicants resulted in increased mortality rates (i.e. time to die) at higher temperatures (the only temperatures that were tested were 35 and 38°C). Exposing the snails to both sublethal level of toxicant and increased temperature resulted in a further increase in mortality rates. Differences between species indicated that one of the species was more tolerant. However, not all toxicants had the same effect.

De Broeck G, De Smet H and Blust R (1995) The effect of sublethal levels of copper on oxygen consumption and ammonia excretion in the common carp, *Cyprinus carpio*. Aquatic Toxicology 32: 127-141. Details of the holding conditions, in standard water, light and temperature regimes are given in the paper. The age of the fish is unknown, but the fish used were all between 15 and 30 grams (unknown sex). Although the paper is specifically about the responses to copper in water, by measuring oxygen uptake and ammonia excretion, baseline data provides information on the response of the fish to low oxygen levels in the water. These fish are oxygen regulators, i.e. they are able to regulate their oxygen consumption despite the environmental oxygen levels, but only up to a level - if the ambient oxygen level drops below this critical oxygen level, the fish no longer regulates its oxygen consumption (becomes stressed?). Five fish were used for each of the trials, with 2 days recovery between trials (under-replicated or psuedoreplication?).

deKozlowski SJ and Bunting DL II (1981) A laboratory study on the thermal tolerance of four southeastern stream insect species (Trichopteraa, Ephemeroptera). Hydrobiologia 79: 141-145. Insufficient information regarding the number of animals exposed. The different species were collected and tested at different times of year: thermal sensitivity may vary seasonally, and comparisons are therefore not realistic; the species are highly seasonal and may not occur together, although not indicated in the study. No indication of the range of temperatures tested in the study. Species tested when the ambient water temperature from which they were collected was warmer than those collected during colder months appeared to have a higher thermal tolerance, although this may be an artefact of the collecting and testing method.

Garten CT Jr and Gentry JB (1976) Thermal tolerance of dragonfly nymphs. II. Comparison of nymphs from control and thermally altered environments. Physiological Zoology 49: 206-213. Compared dragonfly nymphs from thermally altered streams and ponds to those from "control" environments which had not received thermal input. Nymphs from these ponds and streams (7 species in total) were transported to the laboratory and acclimated for at least 48hrs to 1 of 6 temperatures, in stream water: 12, 16, 20, 24, 28 or 32°C. Nymphs (9 for each temperature tested) were then exposed to the test conditions to test for thermal tolerance: measured critical thermal maximum (temperature at which the animals are unable to respond to stimuli) and lethal temperature (all body movements cease and the animals are judged to be dead). The temperature regime and habitat from which they had been removed and the temperature at which they had been acclimated in the lab affected the thermal tolerances of the species: accounted for 81% of the thermal tolerance variation. Nymphs from streams had lower thermal tolerances than those from pond environments. The larger species appear to be more tolerant: the within species body length relationship with thermal tolerance was not examined.

Gray RH (1990) Fish behavior and environmental assessment. Environmental Toxicology and Chemistry 9: 53-67.

Hathaway CJ and Stefan HG (1995) Model of *Daphnia* populations for wastewater stabilization ponds. Water Research 29 (1): 195-208. The paper describes a model of *Daphnia* populations in wastewater stabilization ponds, and no actual data is produced; the data used to produce and simulate the model was taken from various other papers. The model makes various assumptions, most of which do not hold true under normal (ecological) conditions.

Havens KE (1993) Pelagic food web structure in acidic Adirondack Mountain, New York, Lakes of varying humic content. Canadian Journal of Fisheries and Aquatic Sciences 50: 2688-2691.

Korpelainen H (1986) The effects of temperature and photoperiod on life history parameters of *Daphnia magna* (Crustacea: Cladocera). Freshwater Biology 16: 615-620. 5 Clones of *Daphnia magna* were isolated and maintained in the laboratory, in aged tap water. These were then subjected to different light and temperature regimes and mortality rates determined, as well as sex ratios and size at death. There were differences between the clones, as well as between the sexes: males smaller than females. All clones and males and females showed the same reponse to increasing temperatures: the lifespan decreased significantly. The pattern for affect on size of animal at death was not clear: significant differences are indicated, athough no range-finding tests were done to establish where the differences were. The generation time time decreased with a concommitant increase in the intrinsic population rate increase, but with fewer male offspring being produced.

Maltby L (1995) Sensitivity of the crustaceans *Gammarus pulex* (L.) and *Asellus aquaticus* (L.) to short-term exposure to hypoxia and unionized ammonia: observations and possible mechanisms. Water Research 29 (3): 781-787. Used 3 replicates for each of the tests, with 10 individuals per container. The control level was set at 10mg$O_2$/ℓ, the test levels were all below this. The ventilation rate was measured as the mean number of pleopod beats for 2 minutes for 2 trials. Differences in tolerances (mortality) between males and juveniles of the same species and males and juveniles of the 2 different species. The differences in ventilation rates showed similar trends, although the rates of increased ventilation were different (at the same DO levels). At $O_2$ levels below 2mg/ℓ the ventilation rates decreased again, although no indication of why this happened.

Naylor C, Pindar L and Calow P (1990) Inter- and intraspecific variation in sensitivity to toxins; the effects of acidity and zinc on the freshwater crustaceans *Asellus aquatics* (L.) and *Gammarus pulex* (L.). Wat. Res. 24 (6): 757-762.

Nebeker AV (1972) Effect of low oxygen concentration on survival and emergence of aquatic insects. Transactions of the American Fisheries Society 4: 675-679. The paper provides some data for the tolerance levels of other animals to low oxygen levels (not all papers were available for this review): cold-water mayflies and stoneflies cannot tolerate levels below 5mg/ℓ; insects such as mosquitoes can tolerate levels as low as 0mg/ℓ; the mayfly *Hexagenia limbata* was killed at levels of 0.7mg/ℓ; many midge species can tolerate oxygen concentrations as low as 1mg/ℓ; cognisance must be taken of temperature and water velocity. Most animals are able to tolerate short bursts (acute) of low oxygen levels, while little information is available on the long-term effects on growth, survival and reproduction. Used 8 species to test their responses over 96-hrs, tested 4 species to test over 30 days, 4 species were tested for survival to emergence. For the duration of the experiment, the animals were restricted from the water surface (tests showed that access to the water surface increased the LC50, i.e. were able to survive reduced levels of DO). To ensure successful emergence to adult, the mayfly larvae required higher DO levels than the LC50 values resulting from the 96-hr test: for a 50% emergence, the minimum DO levels are 6mg/ℓ (in some cases, the minimum requirement is higher). The paper recommends carrying out a minimum test of egg to egg cycle, to properly ascertain the effects of low oxygen levels on the survival and success of these aquatic insects.

Nordlie KJ and Arthur JW (1981) Effect of elevated water temperature on insect emergence in outdoor experimental channels. Environmental Pollution (Series A) 25: 53-65. Experiments where water temperature was elevated 10°C above the ambient water (control) showed that insect emergence patterns were altered: no guideline limits were established. No indication of the threat or extent of a pollutant of this nature (elevated temperature) on the synchronicity or population success of the insects. (Used a mesocosm experimental channel system, and an ANOVA design experiment).

Price EE and Swift MC (1985) Inter- and intra-specific variability in the response of zooplankton to acid stress. Canadian Journal of Fisheries and Aquatic Science 42: 1749-1754.

Pynnönen K (1995) Effect of pH, hardness and maternal pre-exposure on the toxicity of Cd, Cu and Zn to the glochidial larvae of a freshwater clam *Anodonta cygnea*. Water Research 29 (1): 247-254. Low pH (4.5-5) reduced the viability of the glochidia. (no more useful information could be extracted).

Sherberger FF, Benfield EF, Dickson KL and Cairns J Jr (1977) Effects of thermal shocks on drifting aquatic insects: a laboratory simulation. Journal of the Fisheries Research Board Canada 34: 529-536. Determined the potential effect of thermal plumes on drifting insects, where the exposure time may be brief. Holding condition details are given in the paper: dechlorinated tap water, with sodium thiosulphate added to remove any residual chlorine; pH approximately 7.8. The thermal plume was simulated by exposing the test animals instantaneously to the test temperatures for appropriate times and then gradually reducing the temperature to the ambient (acclimated) water temperature. Postshock observations were carried out for 10 to 53 or 122 days (depending on species) after exposure. Exposure times and thermal tolerances were established by gradual increments in both. Animals were acclimated to different temperatures, and then exposed to a range of increases in temperature. The acclimation temperature affected the thermal tolerances: both in the temperature they could tolerate and as well as the exposure time to those increased temperature. Those acclimated to higher temperatures had higher thermal tolerances (but appeared to be moderated by the months in which the animals were collected?). There did not appear to be an effect on behaviour, but this may have been a result of the experimental design: only animals that had recovered were used in the feeding experiment. Moulting increased up to the thermal tolerance level of 26-29°C for *Isonychia*, afterwhich it decreased again.

Smiley PC Jr and Parsons GR (1997) Effects of photoperiod and temperature on swimming performance of white crappie. Transactions of the American Fisheries Society 126: 495-499. Temperature affected the swimming performance of the white crappie (*Pomoxis annularis*), measured as swimming speed: increasing temperature resulting in increasing swimming speeds; changes in behaviour may have longer-term effects on physiology (growth, reproduction *etc.*) and ecology (ability to escape predators and catch prey *etc.*). Not useful for the review, as the paper did not consider the possibility that fish may be able to move away from undesirable conditions (not part of the scope of the paper), and no limits of water temperature were examined.

Szal GM, Nolan PM, Kennedy LE, Barr CP and Bilger MD (1991) The toxicity of chlorinated wastewater: instream and laboratory case studies. Research Journal WPCF 63: 910-920.

Taylor AC and Funge-Smith SJ (1994) Temperature and salinity effects on oxygen transport by the hemolymph of the freshwater prawn *Macrobrachium rosenbergii* (de Man). Physiological Zoology 67 (3): 639-658.

Toetz DW (1981) Effects of pH, phosphate and ammonia on the rate of uptake of nitrate and ammonia by freshwater phytoplankton. Hydrobiologia 76: 23-26 (appalling paper, not considered useful).

Vijverberg J, Kalf DF and Boersma M (1996) Decrease in *Daphnia* egg viability at elevated pH. Limnology and Oceanography 41 (4): 789-794. Based on other authors' work, most cladocerans have an upper pH limit of 10.5-11.5. *D. galeata* were collected from the lake and reared to F2: these offspring were used in the tests. Standard techniques for food preparation and medium were used: pH 9, temperature 17.5°C. Daphnia were placed individually in 100mℓ of the different pH solutions (9, 9.5, 10 and 10.5), and the test medium was replaced every 2-3 days. The *Daphnia* were measured, and the number of eggs produced and degenerated counted. These values were used in the Euler equation to produce an *r* value, which is the per capita rate of increase for the population. The equation shows the greates impact on the population growth curve at pH levels above 10. It is suggested that these pH effects are direct toxic effect, rather than affecting the food availability to the *Daphnia*.

Zlotnik I and Dubinsky Z (1989) The effect of light and temperature on DOC excretion by phytoplankton. Limnol. Oceanogr. 34 (5): 831-839.

## GENERAL NOTES

- Limited extrapolation information and uncertainty calculations.
- Little South African information.
- Limited extrapolation of single species to populations, other species and ecosystems. Where extrapolation is mentioned, testing is recommended.
- The extrapolation from the laboratory to the ecosystem is a smaller step for system variables than for toxics.
- Salinity: difficult to identify stressor-response relationships. Salinity effects on riparian vegetation is important, and not addressed by the water quality guidelines for the aquatic ecosystem, or the Reserve.
- Ecosystem-type (mesocosm) population toxicity testing needs to be undertaken, and mathematical models used to extrapolate results to field conditions. Exposure models are available for variables such as toxics, nutrients, TDS, DO and pH.

## APPENDIX 4
## LITERATURE REVIEW ON STRESSOR-RESPONSE RELATIONSHIPS - HABITAT:

*Elna Vermaak, IWQS*

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncert. calculation | Level | Type (metric) | |
| Field | Riparian vegetation class | Level of impairment | Reference conditions | Scoring system | No | River reaches | Integrity | Kleynhans, 1996 |
| Field | Fish species | Habitat area (square metres) | Sampling and US environments | Regression equation (graphs) | | Ecosystem | Population (species richness) | Schlosser, 1985 |
| Field | Aquatic ecosystem | Habitat area (square metres) | Sampling and US environments | PHABSIM model | | Ecosystem | Habitat (relationship with streamflow)) | Douglas and Johnson, 1991 |
| Field | Ecosystem | Flow (variability on in-stream habitat) | New Zealand environment | Statistical Analysis | | Aquatic communities | Conditions (univariate analysis of variance) | Jowett and Duncan, 1990 |
| Previous studies | Fish species | Flow (habitat) | Hydraulic parameters | Suitability graphs | | Diversity of habitat | Relationship (suitable habitat & discharge) | James, 1994 |
| Field | Ecosystem | In-stream habitat | New Zealand environments | General rules | | Ecosystem | Conditions (favourability) | Biggs, Duncan, et al, 1990 |
| Field | Macro invertebrates | Flow habitat structure | Hydraulic conditions | Rule | | Invertebrate communities | Population | Lancaster and Hildrew, 1993 |
| Field | Fish communities | Environmental variability (floods & droughts) | Regulation of stream fish assemblages | Regression equations -- - *(a)* | | Stream reaches to represent the range of habitats (pools, raceways & riffles) | Population (fish density) | Schlosser, 1991 |
| Model | Aquatic ecosystem (downstream of impoundment's) | Incoming Water Quality (Precipitation chemistry, basin chemistry, terrestrial vegetation, climatic conditions, anthropogenic impacts) | Limnological Phenomena ( Morphometric parameters, retention time, impoundment age, turbidity currents, drawdown extent, thermal stratification, chemical stratification, edaphic factors, biological activities) | Operational Variables (discharge pattern, release depth) | | Stream, floodplain and estuarine environment (Thermal regime, flow regime, ground water, nutrient levels, substratum type, water clarity, lentic plankton, dissolved salts, dissolved gases, organic detritus) | Biological effects (Riparian vegetation, aquatic vegetation, ecological diversity, biotic productivity, migratory movements, species composition, life-cycle phenomena, trophic structure, potable water, recreation) | Ashton, 1992 |

| INPUT | | | RELATIONSHIP | | | OUTPUT | | REF |
|---|---|---|---|---|---|---|---|---|
| Source | Level | Type (Metric) | Parameters | Form | Uncert. calculation | Level | Type (metric) | |
| Field | Macroinvertebrates (Plecoptera) | *Habitat classes* | Czechoslovakia environment | Ecological indexes, similarity indexes and statistical evaluation | | Community | Population | Helesic & Sedlak, 1995 |
| Field | Macroinvertebrates | Littoral habitats | Flow (changes in discharge) | Classification and ordination | | Segments — *(b)* | Family level presence/absence | Humphries, Davies & Mulcahy, 1996 |
| Previous studies | Fish population | Instream habitat | Flow | Time series | | Habitat area (sq m) | Equalled or Exceeded | Stalnaker & Bovee, 1996 |
| Field | Brown Trout (*Salmo Trutta*) & Atlantic Salmon (*S. Solar*) | Habitat use | Flow | Habitat-hydraulic modelling — (*c*) | | Habitat occupied by fish | Habitat availability | Heggenes, 1996 |

- The Physical Habitat Simulation (PHABSIM) System is a series of computer programs used to implement the Instream Flow Incremental Methodology (IFIM). IFIM was developed to fill a particular need for decision makers in the water resources arena for a quantitative method to assess fish habitat trade-offs against other uses of water. The goal of the method was to relate fish and wildlife values to stream discharge in a manner generally consistent with methods for quantifying other beneficial uses of water.
- The IFIM process includes evaluation of effects of incremental changes in stream flow on channel structure, water quality, temperature and availability of suitable microhabitat in order to recommend a flow regime that will maintain existing habitat conditions.
- Natural resource agencies have attempted to quantify and measure aquatic habitat as a function of streamflow for rivers and streams. This relation is complex and multi-faceted.
- (a) Comparisons of annual changes in species-area relationships for the total community, and for the community with juvenile fish excluded, were used to access the impact of annual changes in juvenile abundance on species richness adjusted for area effects. Specie-area relationships were compared using standard analysis-of-covariance procedures with area as the covariance.
  Species-area relationships (log[species richness] = a + b log[patch area])
  Area of each habitat patch (pool, raceway & riffle) = length x mean width
- (b) Segments was based on changes in sinuosity, discharge, gradient, pool : riffle : run ratios and the magnitude of irrigation abstractions.
- The relationship between flow and the amount of suitable habitat is non-linear. Habitat methods provide information on how habitat changes with flow for instream uses, either biological or recreational. The outcome depends on what species is used or considered and what suitability curves are used. (Jowett, 1997).
- (c) Important physical variables influencing habitat-hydraulic modelling use by brown trout and Atlantic salmon are water depth, water velocity and current shear, substrate particle size and cover.

# PART 3

## SPECIALIST WORKSHOP 2:

## RISK-BASED OBJECTIVES *FLOW* WORKSHOP

This Part contains the output of a workshop concerning stressor-responses relationships, with flow as the identified stressor. The workshop compared two different risk-based approaches for altering flow requirements for various Ecological Management Classes, and looked at the methodology for determining the water quantity Reserve. A preliminary attempt was also made to define different levels of flow-related stress for instream riverine fauna.

## TABLE OF CONTENTS

## INTRODUCTION

This workshop was held as a result of the outcome of the risk-based objectives (RBO) workshop (Specialist Workshop 1) held in August 1998, where a number of stressor-response relationships had been discussed. Flow was identified as a stressor which could be investigated in some detail almost immediately, as the DWAF In-stream Flow Requirements (IFRs) team was already implicitly following a risk-based approach, when determining water quantity requirements during flow estimates and Ecological Reserve determinations. The workshop was attended by the following delegates:

Prof Denis Hughes, IWR
Prof Jay O'Keeffe, IWR
Ms Delana Louw, IWR Environmental
Mr Nigel Kemper, IWR Environmental
Mr Drew Birkhead, Streamflow Solutions
Ms Rebecca Tharme, Southern Waters
Prof Kate Rowntree, Geography Department, Rhodes University
Dr Neels Kleynhans, IWQS
Ms Christa Thirion, IWQS
Mr Niel van Wyk, DWAF
Mr Sebastian Jooste, IWQS
Dr Patsy Scherman, IWR

At the first RBO workshop, flow was discussed in terms of setting water quantities for Environmental Management Classes (EMCs) during Reserve assessments, and how to alter flow requirements for various EMCs. Two scenarios were presented, which could be summarized as the 'less depth' vs. 'less frequency/assurance' scenarios (see Specialist Workshop 1 report, Part 2). *Note that frequency and assurance are used interchangeably in this document.*

After the first specialist workshop, the two methods were provisionally tested at the Crocodile River IFR workshop held during 1998. This RBO flow workshop therefore presented an opportunity for further testing of the two methods, using the following approach:

- Choose a range of rivers with a range of hydrological variability for which the necessary IFR data are available.
- Compare the hydraulic and hence ecological consequences of applying both methods.
- Try to determine which, if either, of the methods should be generally adopted, or if some compromise or alternative method could be developed.

### Workshop objectives

- To determine whether a risk-based approach is acceptable to use when setting the quantity Reserve, and when extrapolating to EMCs other than that for which the Ecological Reserve was set.

- A risk-based approach could be coupled to different assurances of maintenance flows for different EMCs (the '*less frequency/assurance*' method), or by determining motivated higher/lower flows for different EMCs (the '*less depth*' method).

  'Less frequency/assurance' therefore refers to maintaining the depth of the maintenance flow, but altering its assurance or frequency, while 'less depth' refers to a change in maintenance flow (volume) per EMC.

*Note 1: When referring to depth, note also referring to **velocity** and **wetted perimeter**.*

*Note 2: Both approaches are essentially risk-based. The feasibility of using either approach in a risk-based context is determined by the links that can be made between changing flow levels and stress on riverine biota.*

Appendices 1, 2 and 3 form the documentation of the RBO flow workshop. Appendix 1 is the workshop starter document, which details background information on the 'less depth' vs 'less frequency/assurance' determination methods, as well as the requirements for a risk-based approach. Appendix 2 is the IFR information to be used in the workshop, while Appendix 3 is the workshop output as regards the testing of the two methods using IFR data.

**Workshop outcomes**

At present, flow assessments are linked to hydrological indices only, and are not linked to ecological effects. However, a description of risk must contain three elements: A stressor (e.g. flow), an end-point or effect (e.g. on the biota), and the probability of an effect. Attempts to link changing flow scenarios to stress on biota will probably be at the level of the habitat, i.e. changing flows will directly affect habitat conditions, which will effect the biota.

***A comparison of 'less depth' vs 'less frequency' methods***
Although the 'less frequency (or assurance)' method may be better suited to management objectives, the results of Appendix 3 could not show a generic preference of one method over the other for the setting of maintenance flow for different EMCs. It is possible that each case should be handled independently, the suitability of each method assessed, and the appropriate method selected. Although useful, the assessment of both methods would be costly and impractical. It was suggested that a *decision framework* be developed which could guide scientists in their assessments.

***Stressor-response relationship: flow and biota***
This section of the workshop resulted in much fruitful discussion around determinations of 'flow stress', and therefore the risk of changing flows to biota. Although the original thinking was stress in terms of changing habitat, there was general recognition that IFR workshops currently proceed with implicit assessments of stress to biota. A method is therefore required to utilize this information, and generate nominal quantitative information from qualitative specialist information. The following documents by Hughes and O'Keeffe respectively, are a first attempt to develop a framework for

determining the water *quantity* Reserve, and defining different levels of flow-related stress for instream riverine fauna.

## DOCUMENT 1
## A FRAMEWORK FOR THE METHODOLOGY OF DETERMINING THE WATER QUANTITY RESERVE

*Denis Hughes, Institute for Water Research, Rhodes University*

### INTRODUCTION

The ideas expressed in this document represent my interpretation of various developments that have taken place over the last 12 months within some IFR workshops, meetings related to the development of the 'Intermediate Determination' methodology and meetings related to the development of methods to specify initial low-confidence estimates for the country as a whole (the 'Desktop Estimate'). This document should be considered as a first attempt, is therefore far from complete and only superficially explains some of the new concepts that are being considered by some of the specialists currently working in the field. It is therefore offered as a working document, and to generate ideas, suggestions, discussion and criticism by the community of specialists involved in IFRs and the Reserve. This research has been updated and expanded in the following two documents:

Hughes, D.A (1999). Hydrological information requirements for the determination of the Ecological Reserve for South African rivers. Ninth South African Hydrology Symposium, 29-30 November 1999, Western Cape.

Hughes, D.A. and Münster, F. (in press). Provision of hydrological information and a hydrological based decision support system for the determination of the water quantity component of the Ecological Reserve for rivers. Report to the Water Research Commission by the Institute for Water Research, Rhodes University.

Background information can be found in:

Hughes, D.A. (1999). Towards the incorporation of magnitude-frequency concepts into the building block methodology used for quantifying ecological flow requirements of South African rivers. Water SA 25(3): 279-284.

### GUIDING PRINCIPLES

The following principles were adopted in drawing up the framework:

*Principle 1* We need a range of approaches available for quantifying the Ecological Reserve, which should range from initial low-confidence estimate methods through to methods that give estimates of the highest possible confidence. Given that this is the case, it is important that they should be based on the same basic concepts. The reasoning behind this principle is that developments in one area can contribute to strengthening others.

the system in conjunction with the information supplied by the hydraulic specialist. The specialists should be encouraged to consider (in a qualitative way) to what extent their component of the aquatic system will be 'stressed' under different flow conditions.

Step 4 With the assistance of a facilitator define approximate 'stress'/flow relationships for individual components of the system (fish, invertebrates, riparian vegetation, etc.) at an appropriate level of quantification (see Figure 1.1A). These would be defined for (probably) two key months of the year (wet and dry seasons). 'Stress' can be interpreted in a flexible manner and the quantification will necessarily be on a nominal scale (e.g. 10 = highly stressed, 5 = moderately stressed, 0 = not stressed). Generate a combined stress/flow relationship for the system as a whole which is likely to be an envelope curve (Figure 1.1B).

Step 5 Use the time series of flows for natural and present day (or historical) flow regimes and combine them with the stress/flow curves to determine baseline indices or profiles of stress/duration. It has been pointed out by Jay O'Keeffe that a single stress value is not sufficient, as aquatic systems need to be under stress some of the time to be able to function correctly. A 'profile' approach (along the lines of Figure 1.2) seems to be the best suggestion at this stage. The profile will probably have to consist of several items of information including a type of duration curve of 'stress' (i.e. % time certain levels of stress are equalled or exceeded within the whole time series), as well as run-analyses of stress (i.e. lengths of time that the system is at or above certain stress levels). Figure 1.2 provides some further details, but much work is needed to clarify these concepts. From a practical point of view, if the ecological specialists can define the stress curves, it is relatively easy to develop software to integrate such curves with the hydrological data and generate stress profile information (the approaches are similar to methods already incorporated into the HYMAS software).

Step 6 The IFR model (that generates a time series of IFR requirements based on a set of rules that determine how often maintenance and drought flows should occur), or a similar technique, could then be calibrated by the specialists (facilitated by the hydrological specialist) to generate several possible time series of modified flows. From these time series, new 'stress' profiles could be generated (as in Figure 1.2) and compared with the natural profile to try and optimise the IFR operating rules. It is difficult to suggest at this stage what would be involved in the optimisation - a possibility is to try and obtain a stress profile which is as similar as possible to the natural stress profile. Inevitably, it will involve an iterative process using the knowledge of the specialists.

Step 7 If it is considered necessary to offer alternative scenarios of IFR requirements (e.g. for EMCs lower than the preferred one), the process in Step 6 can be repeated for various alternative approaches to reduce the overall long-term IFR volume requirement. Thus, further stress profiles can be generated which can be evaluated to determine the best option. For example, to obtain a C class IFR having already set a B class IFR, there have been several discussions about whether to reduce maintenance flows and retain the same assurance rules ('less depth' method) or whether to retain the same maintenance flow but specify that it should occur with a lower assurance ('less frequency/assurance' method) (and there are other options as well). The use of stress profiles could assist in resolving such issues.

Step 8 Pass on the assurance rules and/or representative time series of IFR requirements to the water resource engineers, so that they can design the necessary developments and account for the IFR in a way which is compatible with the needs of the river as defined by the ecologists.

**Low-confidence approach** (Desktop Estimate type approach)

Step 1 Use existing and developing methods to define the preferred future EMC based on such as the eco-region approach.

Step 2 Use a generic relationship between annual IFR totals (baseflow and total flow, for example) and some hydrological and/or ecological indices for different EMCs. Figure 1.3 provides an example of an existing relationship that has been developed from an integration of past IFR workshop results. It is very simplified and neglects ecological considerations related to differences that may be related to eco-regions, position in the rivers longitudinal profile, nature of the river bed, etc. It is, however, a starting point that can now be improved as more information becomes available.

Step 3 Distribute the annual values into monthlies using regionalised generic seasonal distributions.

Step 4 Use generic assurance rule curves (based on regionalised flow regime characteristics and additional ecological factors - Figure 1.4) to determine the frequency with which defined flows (maintenance and drought) are expected to occur within the modified flow time series, and generate such a time series.

Step 5 Make use of generic stress/flow relationships combined with the natural and modified flow regimes to compare stress profiles, and where necessary modify the assurance rule curves to obtain a more balanced (in terms of stress) modified flow regime. Generic stress/flow regimes might be based on channel morphology and regional ecological factors, but much thought needs to be given to this aspect. The approach developed for the Desktop Estimate at the IWR, Rhodes University, makes use of monthly time series of flow and consideration needs to be given to how these could be integrated with stress/flow relationships, compared with the high-confidence approach suggested above, using daily flow data.

Step 6 If the Desktop Estimate were to be used to specify requirements for a different EMC, then similar procedures as in step 5 (above) and step 7 of the high-confidence approach could be adopted.

Step 7 Pass on the assurance rules and/or representative time series of IFR requirements to the water resource engineers, so that they can design the necessary developments and account for the IFR in a way which is compatible with the needs of the river as defined by the ecologists.

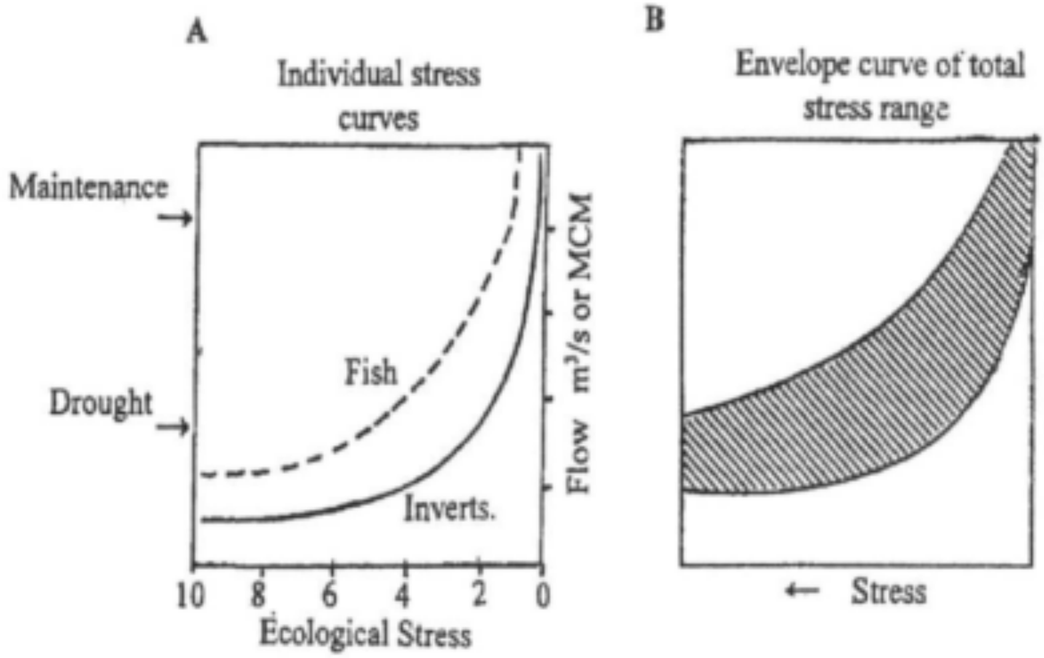
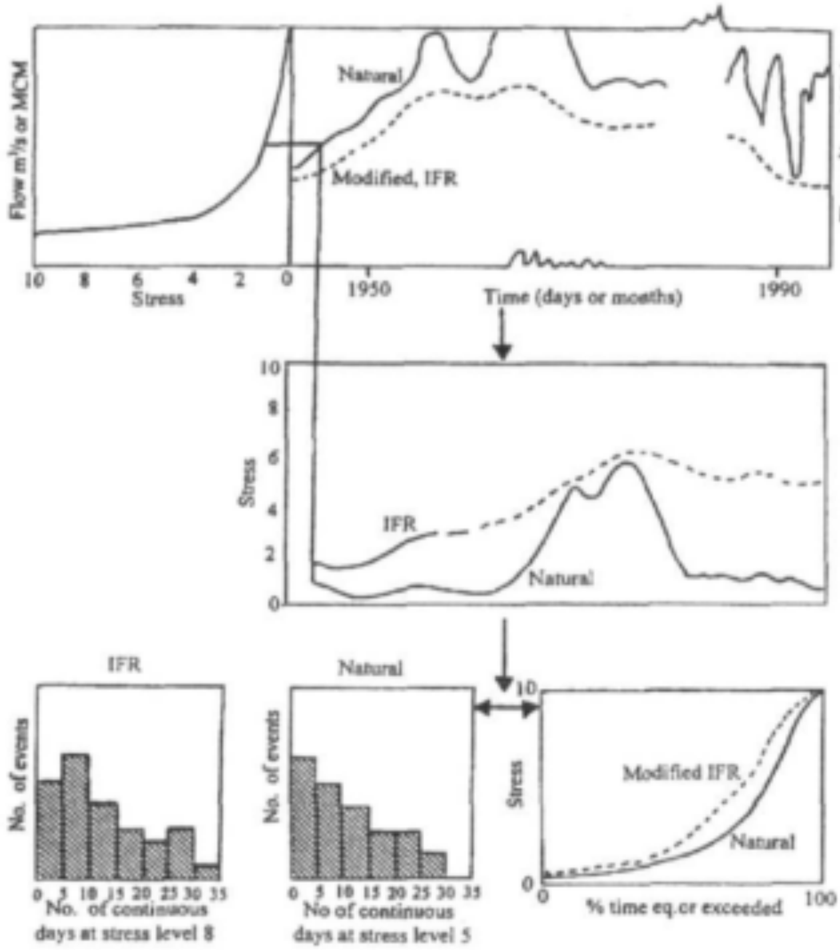



**Figure 1.1** Stress/flow relationships

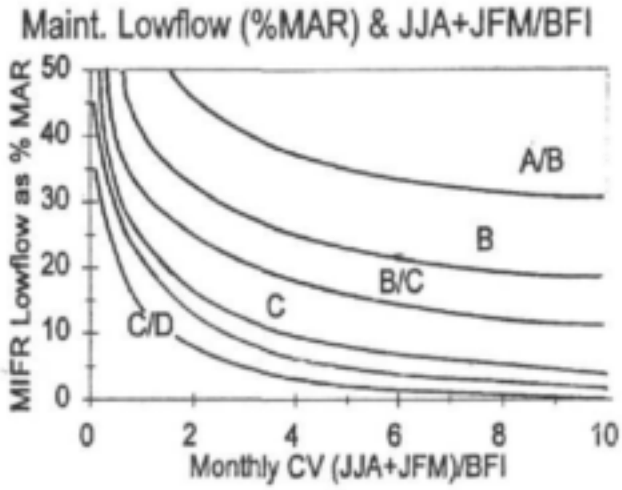
It is proposed that these could be defined from the specialists qualitative ideas about how the system operates under different flow conditions. It is suggested that these qualitative ideas are already being used to determine maintenance and drought flows and 'all' that is required is to express them in some nominal quantitative format. The lines representing the relationships could be drawn with a fine pen, where specialists are confident about their information, or with a wide brush, where there is more uncertainty. In putting together the relationships for the system as a whole (B), an envelope curve is likely to result - it is also possible that different importance weights could be given to different components. One of the perceived difficulties in carrying out this is process is to separate out the 'stress' caused by different flow conditions and the levels of stress resulting from those conditions occurring for different durations.




These lower graphs (plus run-analyses for other stress levels) = stress profile.

**Figure 1.2** Combining stress/flow relationships with flow time series to generate stress profile information.

The Figure illustrates that a stress/flow curve can be used with a flow time-series to generate a time series of 'instantaneous' stress which can then be analysed to generate a duration curve of 'instantaneous' stress, as well as histograms of the number of occasions (events) when defined stress levels were exceeded for specific lengths of time (normally referred to in hydrology as run-analysis or spell-analysis). To facilitate the use of such an approach it will be essential to have available computer software that is transparent in what it is doing and that produces clear and understandable output (mostly graphical).



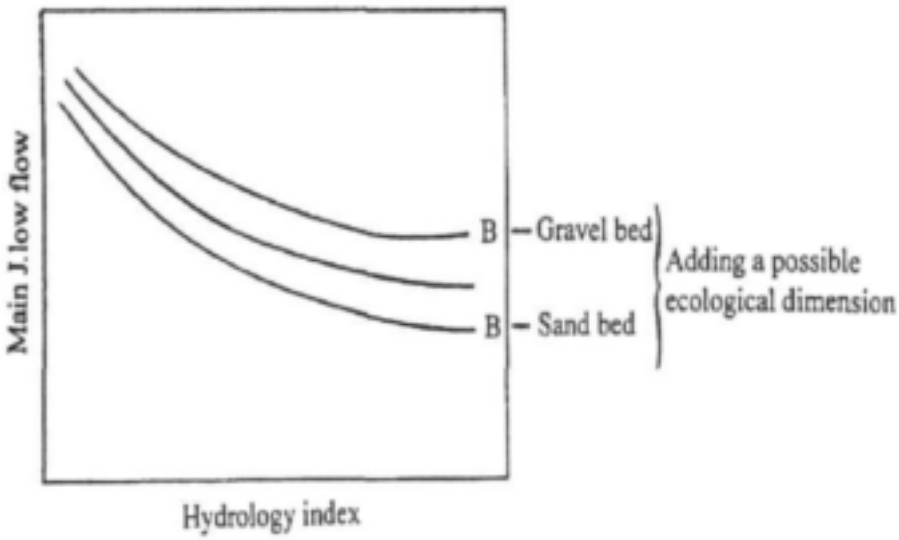




**Figure 1.3** Relationships between annual IFR requirements and hydrological indices.

The deficiency of this existing relationship is that it ignores purely ecological effects (i.e. those unrelated to flow regime characteristics). Frauke Münster, previously of the IWR, Rhodes University, has been working on a WRC-funded project to try and find ways of incorporating some of these effects. One approach that she has adopted is to try and integrate any qualitative ideas that other specialists (who have worked extensively on aquatic ecological problems) might have and to see if they can be put into a crudely quantitative context.

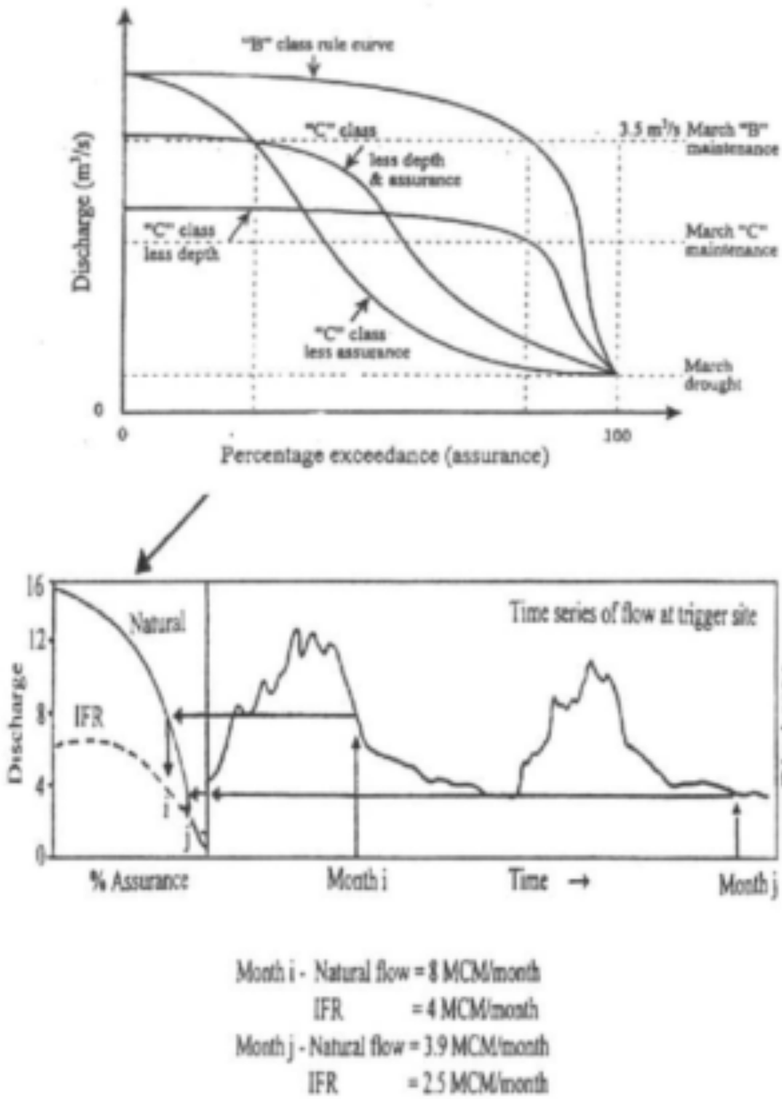



**Figure 1.4** Generic assurance rule curves.

The concept is that for any particular day or month, the flow at a reference or trigger site would be identified and the % position of that flow on the duration curve identified. The same % value would be used to determine the flow that is required in the modified (IFR) time series. This approach is currently being used in the Desktop Estimate approach to generate a time series of monthly flows. The rule curves are also being used within the Department of Water Affairs and Forestry's Yield Model to specify IFR requirements during basin-wide water resource development assessment studies.

## DOCUMENT 2
## POSSIBLE DEFINITIONS FOR DIFFERENT LEVELS OF FLOW-RELATED STRESS FOR INSTREAM RIVERINE FAUNA

*Jay O'Keeffe, Institute for Water Research, Rhodes University*

### INTRODUCTION

This is a first attempt to define a consistent qualitative description of stress levels which could be used to construct stress curve relationships at particular sites. The definitions apply to instream fauna (and therefore separate ones would have to be defined for e.g. riparian vegetation), and are calibrated on organisms that would require flowing water conditions for optimal habitat.

Stress is therefore seen only in terms of reduced flows, and loss of hydraulic habitat. It is acknowledged that many species thrive in stagnant water, and the stress levels described would not apply to them specifically, but to the instream community as a whole, mainly in terms of loss of local (and probably temporary) reduction in species abundance and diversity. Obviously there is a time dimension to stress (e.g. is a stress level of 6 for 3 weeks worse than a stress level of 5 for 4 weeks?). This has been acknowledged in the descriptions to some extent, but is mainly taken into account when the stress curves are related to hydrological time series, to define stress profiles (Hughes, Document 1), by calculating the frequency and duration of different levels of stress.

NB: This stress index refers only to base flows at this stage.

Assuming a dimensionless index of 0 to 10, where:

- 0 = No stress, flows provide a complete range of preferred velocities and depths over a variety of available substrate types, all available biotopes inundated. Extensive availability of critical biotopes.
- 1 = Slight reduction in the abundance of critical habitats. No species will be stressed unless population growth outstrips habitat availability.
- 2 = Progressive reduction in the abundance of critical habitats, but optimum hydraulic habitat conditions are still available in some areas of the site.
- 3 = Further reduction in the abundance of critical habitats, leading to a shortage of optimal areas for critical rheophilic species.
- 4 = Reduction in the abundance of critical habitats reaches the point at which some life stages of critical rheophilic species may not be viable if these conditions persist for several months.
- 5 = Moderate loss of abundance of critical hydraulic habitats. Species with flow-dependent breeding requirements (including habitat for mating, eggs, larvae or juveniles) will not breed, but should survive indefinitely.
- 6 = Only the maximum depths and velocities provide refuge habitat for critical rheophilic species
- 7 = Most of the available habitats become suboptimal due to shallowness, and lack of high flow velocities. Critical rheophilic species will only survive over the short-term (three months).

8 = Critical habitats are reduced to remnant refuges, unsuitable for all except small and hardy rheophilic species
9 = Only stagnant water conditions remain, except perhaps for trickles of water between pools. Local extinction or emigration of rheophilic species, hardy species survive in refuge pools.
10 = Absence of all surface water. Local extinction or emigration of all aquatic species, except those with specialist survival behaviour (e.g. burrowing in the substrate, aestivation, cryptobiosis etc.) due to lack of hydraulic habitat.

Hydraulic habitat - refers to the combination of water velocity, water depth, substrate type and cover which combine to make up a particular type of biotope (e.g. riffle, run, pool, backwater, marginal vegetation in current etc.)

Critical hydraulic habitat - refers to the type of habitat at a particular site which will be most at risk as flows are reduced. Normally riffle/rapid or marginal habitat.

## EXAMPLE:
## THE CONSTRUCTION OF A STRESS CURVE RELATIONSHIP FOR BENTHIC INVERTEBRATES IN THE MIDDLE CROCODILE RIVER (MPUMALANGA)

This example uses hydraulic information from the Crocodile River pilot study (Nov. 1997) on PIFR Site 1. The cross-section and hydraulic relationships used are illustrated in Figure 2.1. It is important to note that the velocities are expressed as averages, and there is no information as to the maximum velocities available at any flow.

**Flow requirements for invertebrate communities**

Information on the invertebrate communities has been extrapolated from a detailed study of hydraulic requirements carried out on the Sabie River (Weeks *et al.*, 1996). In summary, the findings were that:

- The marginal vegetation biotope contained the highest number of taxa (189) and more taxa (24) were confined to this biotope than to others. Riffles contained a slightly lower diversity (178 taxa) of which 13 were found nowhere else. These were therefore the critical flow and water-level related biotopes to be considered.
- 11 taxa common during wet conditions disappeared from samples during drought conditions. 6 of these were Trichoptera or Ephemeroptera, and these two groups are used here as the indicator groups for hydraulic requirements.
- Both groups included species which could tolerate low flow conditions, and those which required higher flows. Both groups showed highest densities and number of taxa at depths between 0 and 0.3m, but large numbers of species (at lower densities) occurred at 0.5m or more. The Ephemeroptera were scarce or absent only at very low current speeds, but the highest number of taxa occurred at between 0.1 and 0.6 m $s^{-1}$, and abundance was highest at current speeds in excess of 1 m $s^{-1}$. For the Trichoptera, the number of taxa was very low at speeds of less than 0.1 m $s^{-1}$, and increased with increasing velocity. Trichopteran abundance was maximum at velocities between 0.6 and 1.0 m $s^{-1}$.

- During extreme drought conditions (equivalent to stress level 9 above) the invertebrate communities of the Sand River were reduced in abundance by an order of magnitude and in number of taxa they were halved compared with wet conditions.

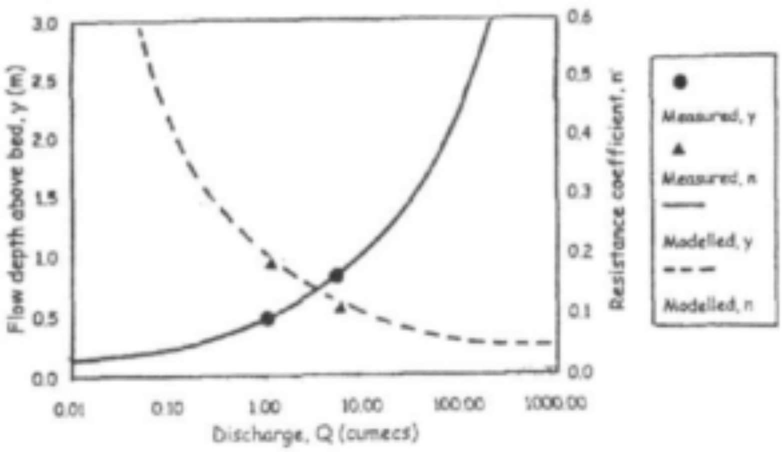
**Relationships between flow and stress for invertebrates at Crocodile River PIFR Site 1 (Illustrated in Figure 2.2)**

- At discharges above 10 $m^3 s^{-1}$ the average velocity is above 0.6 $m s^{-1}$, and average depth is >0.6m, indicating an abundance of optimal habitat for invertebrates, and therefore very low stress. (In the Sabie River, abundances and number of Trichopteran taxa increased at velocities higher than 0.6 $m s^{-1}$, so a stress level of 1 might be justified).
- At discharges of <5 $m^3 s^{-1}$, the wetted perimeter is slightly reduced, average depth is <0.5m, and average velocity is 0.4 $m s^{-1}$. There is a slight reduction in the abundance of critical habitat, but no threat to any species, indicating a stress level of 2.
- At discharges of <3 $m^3 s^{-1}$, maximum depth is still >0.6m, but average depth is <0.4m, and average velocity is 0.3 $m s^{-1}$. There is still water in the marginal habitats, and optimal conditions are still available, but at reduced abundance. This equates to a stress level of 3.
- Wetted perimeter decreases sharply at discharges below 1.5 $m^3 s^{-1}$, average depth is down to 0.3m, and average velocity is 0.23 $m s^{-1}$. Some loss of the important marginal habitat, and a moderate loss of depth requirements for some species indicates a stress level of 5.
- At discharges of <0.4 $m^3 s^{-1}$, maximum depth is 0.4m, average depth is <0.2m, and average velocity is 0.1 $m s^{-1}$. Most of the habitat is sub-optimal, equating to a stress level of 7.
- At discharges below 0.2 $m^3 s^{-1}$, maximum depth is 0.2m, and the effective channel is confined to about 6m on the right-hand bank. Average velocity is <0.2 $m s^{-1}$. These conditions will result in the disappearance (at least temporarily) of the flow sensitive species, and equate to a stress level of 8.
- Stress levels of 9 and 10 would only occur if flow stopped (9), or if surface water disappeared (10).

Crocodile River PIFR

Site 1

Rating : Q = < 0.11 cumecs
y = 0.797Q**0.410
Q > 0.11 cumecs
y = 0.299Q**0.415



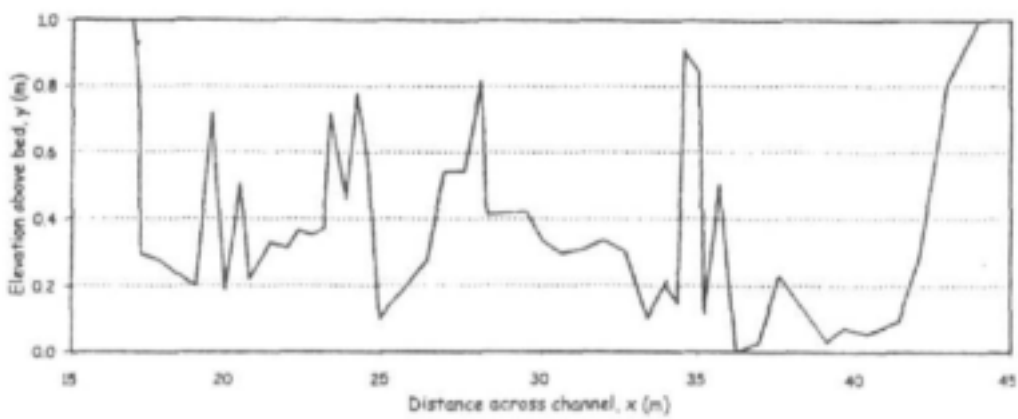
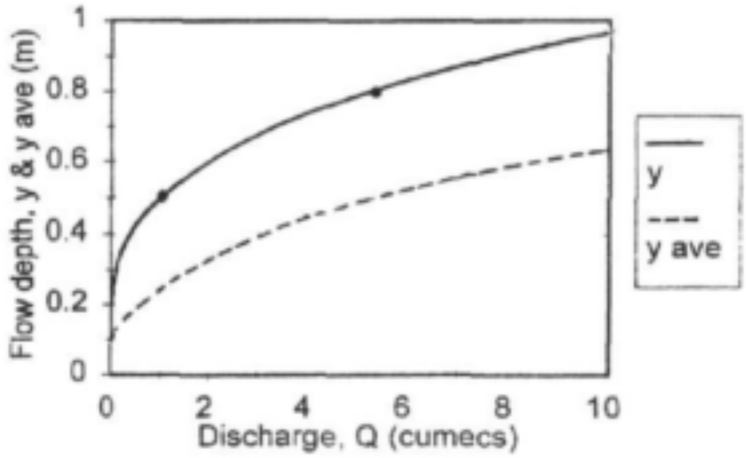
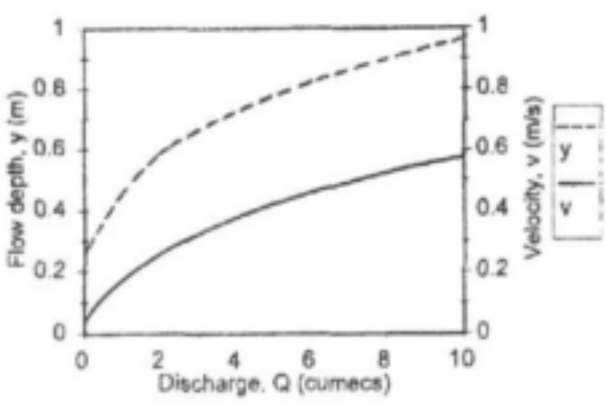




**Figure 2.1** Hydraulic transect and relationships for PIFR1, Crocodile River.

Crocodile River PIFR

| | |
|---|---|
| Site | 1 |
| Flow | 0 - 10 cumecs |



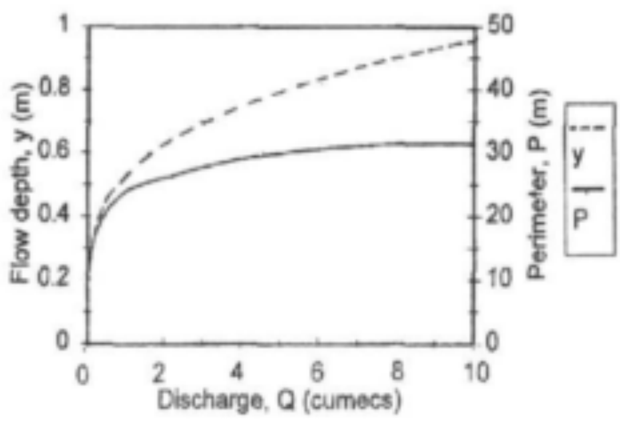




**Figure 2.1 (cont.)** Hydraulic transect and relationships for PIFR1, Crocodile River.

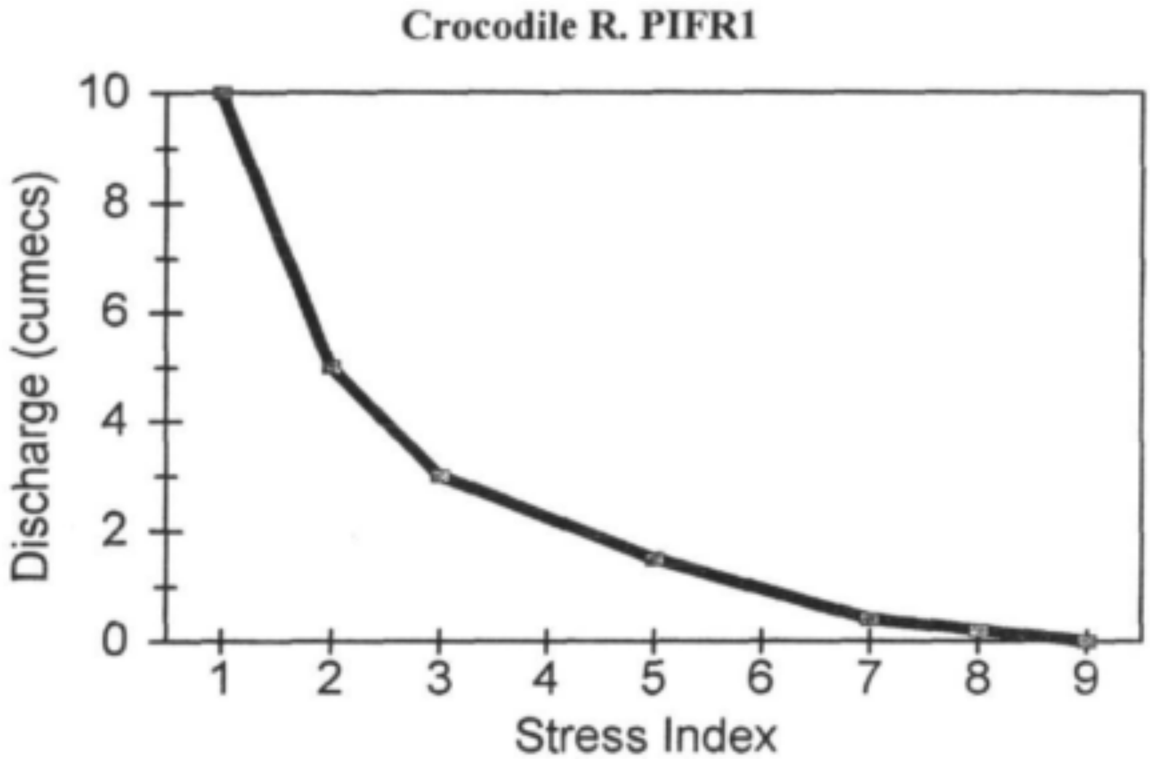



**Figure 2.2** The relationship between flow and stress for invertebrates in the middle reaches of the Crocodile River, Mpumalanga.

## APPENDIX 1
## RBO FLOW WORKSHOP: STARTER DOCUMENT

*Louw, Tharme, Hughes, O'Keeffe, Birkhead*

### 1 INTRODUCTION

### 1.1 IFRs and assurance of maintenance vs drought flows

During the past year, the assurance of maintenance and drought flows were determined **up front** during IFR specialist meetings. A quantitative definition of maintenance and drought flows were therefore determined and all specialists are aware, when determining the IFRs, of how often the flows that they are recommending will occur. At present, no ecological basis exists for determining the % assurance. On the basis of the hydrological characteristics of the river in question, the hydrologist recommends a certain shape (see Figure A1.1) of the flow duration curve and, accordingly, the % assurance of maintenance and drought flows. *Note: This refers only to the* ***baseflow*** *component of the flow regime, not the higher flows (floods).*

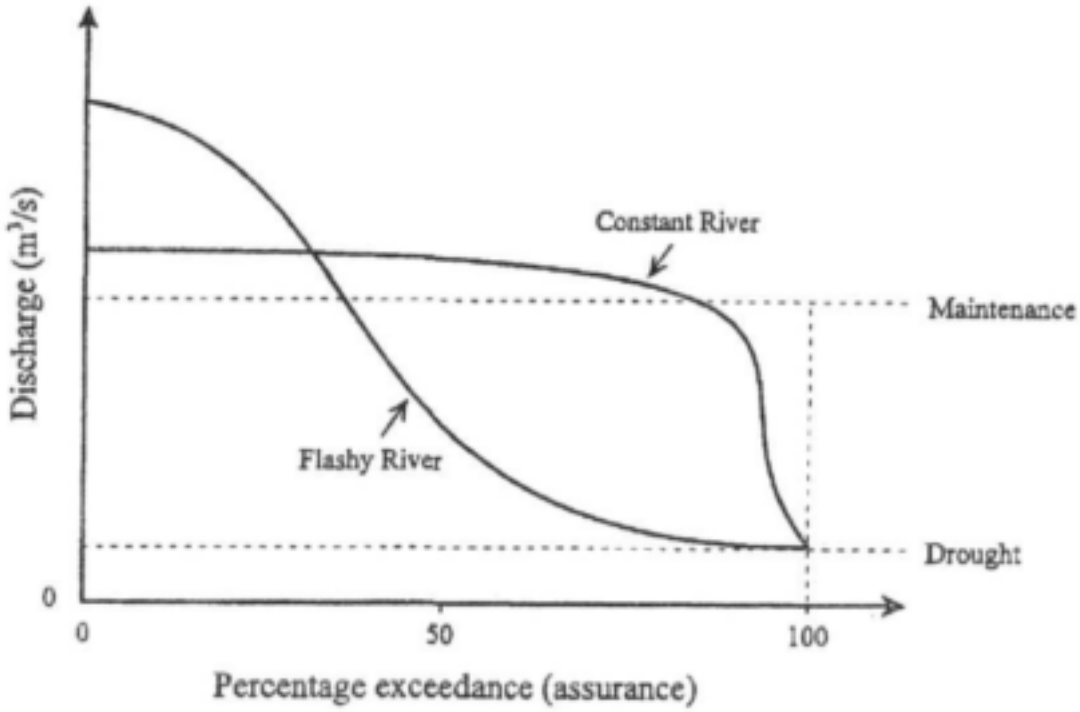



**Figure A1.1** Duration curve for a flashy river vs. a constant river

## 1.2 Mhlathuze River Ecological Reserve (quantity) study

During the Mhlathuze study, the consultants were requested to determine the quantity Reserve for Ecological Management Classes (EMC) either higher or lower than the EMC selected and for which the IFR was determined. Two options of how (in this case degrading, i.e. lowering the EMC) this could be undertaken were put to the specialists:

- Change the percentage assurance of maintenance flows, i.e. the flow rates specified for maintenance flows stay the same, but the frequency changes with different EMC classes. ('less assurance' scenario). The assurance stays the same for maintenance flows, but the discharges and associated hydraulic parameters are decreased ('less depth' scenario).

The two scenarios are illustrated in Figure A1.2 (which is not specific to the Mhlathuze).

In the case of the Mhlathuze River, the 'less assurance' scenario was accepted. The reasoning was the following:

The Mhlathuze at Site 3 is a sand bed river with characteristically slightly deeper flow against one of the banks. The most important motivations supplied for flow during e.g. the dry months were that there should be enough depth in the deeper channel for fish passage and that the deeper flow must be against one of the banks so that the marginal vegetation can be utilised.

In this case, a minimum depth was specified for the maintenance flows that occurred 60% of the time. According to scenario one, a lower EMC would require less depth and a motivation should be coupled to this. However, any less depth will not accommodate fish passage for large fish and this would mean that no ecological motivation can be coupled to these flows as these lower flows would have limited ecological functioning.

However, following the 'less assurance scenario', the same depth and the same motivation can be used. The flow would however just occur less often (30% instead of 60% in the case of the Mhlathuze River). No motivations need then be supplied, only the consequences of supplying the recommended IFR less often.

## 1.3 Crocodile River

As part of the process to determine rapid methodologies for setting the Ecological Intermediate Determination, a pilot study was held on the Crocodile River. The same situation arose where it was required to provide flows for classes other then the EMC used to set the Intermediate Determination. In this case, it was decided to test the two scenarios and a hydrological extrapolation was undertaken to provide a flow duration curve for both scenarios for a C class river (the recommended EMC for which the Reserve was set was a B class). The flows occurring 5%, 50% and 70% of the time were provided and converted to hydraulic parameters of depth, wetted perimeter and velocity (see Table A1.1). A detailed discussion regarding the process was followed, including the ecological consequences of these two scenarios, and the reasons for selecting the 'less depth' scenario is discussed later in this document.

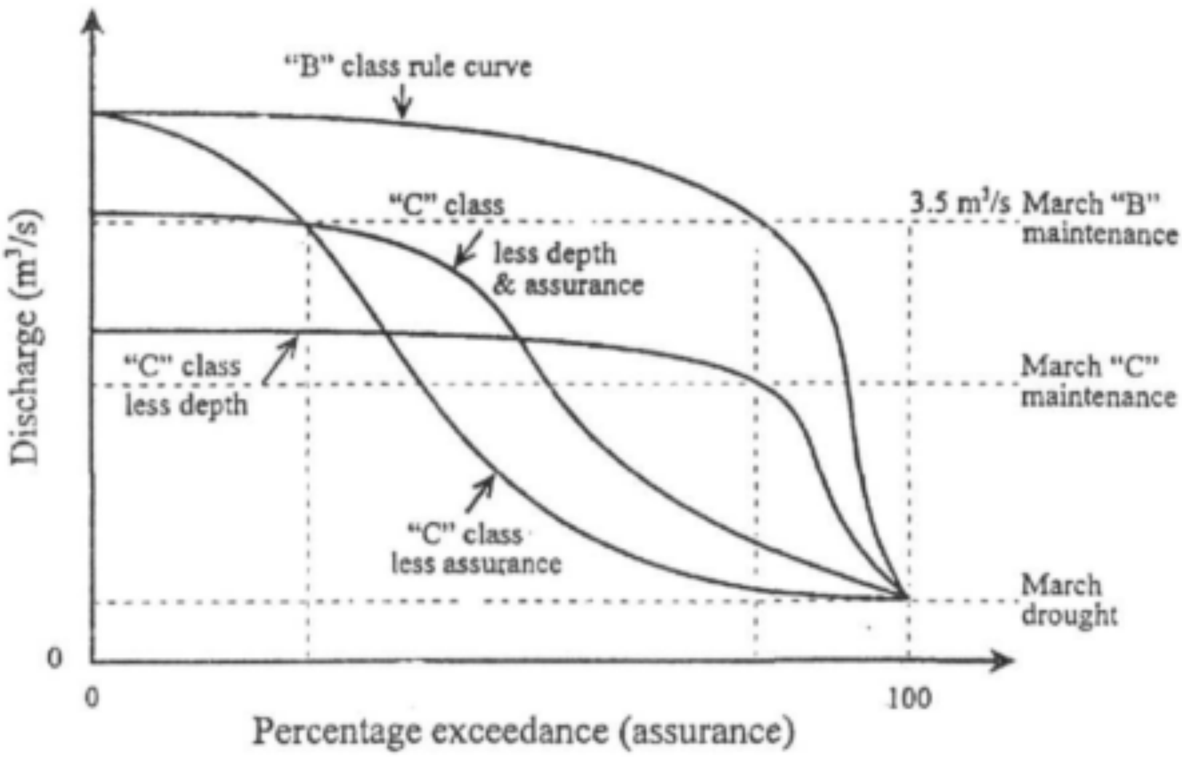



Figure A1 2 Illustration of the 'less assurance' vs. the 'less depth' scenario.

## 2 Mhlathuze vs. Crocodile Rivers

The difference in approach selected by specialists for the Mhlathuze and Crocodile Rivers was discussed during the Crocodile River study. It was established that a preferred approach can only be decided upon if the approach followed for the Crocodile River was applied to various other rivers. These rivers should be hydrologically different to enable us to see whether there is some trends that could be linked to hydrological or biophysical characteristics of the specific river.

The rationale that could indirectly be influencing the different approaches could include some of the following factors:

- The Mhlathuze River is a sandbed river and the Crocodile a pool/riffle river. Different channel forms might be appropriate for different scenarios. The Mhlathuze River was a 'real' study and motivated answers had to be provided during the workshop. As the 'less depth' scenario could not be quantified and motivated for, the 'less assurance' scenario was the most appropriate. The Crocodile River study was a pilot study and this specific discussion was aimed at finding a preferred approach; no definite motivated answers were required. The Mhlathuze River is not perceived as a river in good condition, or a particularly diverse river. The Crocodile River however is perceived as very diverse, in a good condition and ecologically important. Therefore, some emotion could play a role whereby the 'less depth' scenario would seem to be preferable as the flow that would be experienced 70% of the time is higher under this scenario. Therefore, a different class of river could require a different approach. It is easier to quantify the effects of a loss of hydraulic parameters such as depth, than to describe what the effects will be of a loss of hydraulic parameters for only part of the time, i.e. decrease of assurance of specified flows.

## 3 Objectives of the RBO flow workshop

The objective of the workshop can be summarised as follows:

*To determine whether a risk-based approach is acceptable to use when setting the quantity Ecological Reserve and when extrapolating to other EMCs than for those for which the reserve was set.*

*A risk-based approach could be coupled to different assurances of maintenance flows for different classes ('less frequent' scenario), or to determine motivated lower / higher flows than those recommended coupled to different classes ('less depth' approach). The workshop will investigate the results of both these approaches for the existing IFR sites, to determine a suitable scenario or a combination of both scenarios* (The workshop process is illustrated in Figure A1.3).

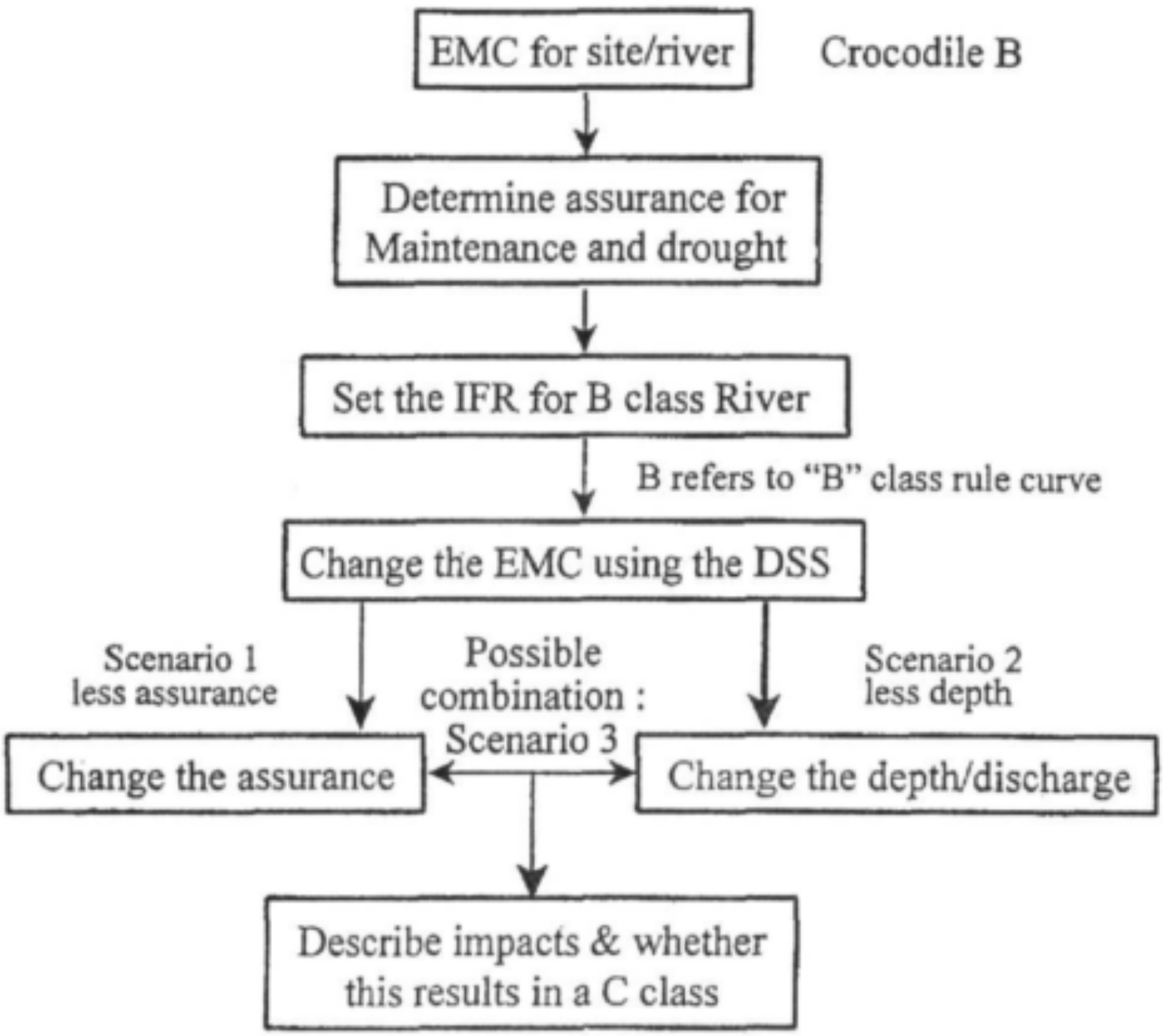



**Figure A1.3** Process followed for evaluating the two scenarios.

## 4 WHY A RISK-BASED APPROACH?

The reasons for adopting a risk-based approach are as follows:

- This approach has been developed and used over the past year, and is a concept embodied in the EMC classification framework (i.e. higher risk of ecosystem failure for lower EMC) (see discussion below). It also maintains the ecological reasoning behind recommendations for particular flows, and is consistent with the management flow concepts for users (e.g. irrigation/industrial water requirements are expressed in terms of % assurance).

The EMC class descriptions are directly linked to risk, e.g. a C class should have a moderate risk of modifying the abiotic template and exceeding the Resource Base may be allowed. Risks to the well-being and survival of intolerant biota (depending on the nature of the disturbance) may generally be increased with some reduction of resilience and adaptability at a small number of localities. The higher the class therefore, the smaller the degree of risk. When an IFR is described for a B river, the flows should constitute a flow regime that has a minimal risk of allowing the river to drop a class. The Resource Quality Objectives for the B class River must then also link to risk. This workshop is therefore seen as being in line with the WRC-funded RBO project.

## TEST CASE:
## AN EVALUATION OF TWO PROPOSED METHODS FOR IDENTIFYING MAINTENANCE IFRs FOR VARIOUS ENVIRONMENTAL MANAGEMENT CLASSES FOR USE IN THE HYDROLOGICAL DECISION SUPPORT SYSTEM

*Rebecca Tharme, Southern Waters, University of Cape Town*

### Introduction

Two main methods of identifying the maintenance IFRs for various environmental management classes (EMCs) were identified and tested:

**Crocodile Method 1 (C1)**: The maintenance IFR flows and percentage assurance rules are set for the identified EMC for the river and then the assurance rules for meeting the maintenance IFR are changed for the different classes

**Crocodile Method 2 (C2)**: Set individual maintenance IFR flows for each EMC and use the same percentage assurance rules as for the identified EMC.

### Background

As a result of the pilot testing of the Intermediate Determination on the Crocodile River, the river was assigned an EMC status of B Class. The preliminary maintenance IFR was derived on the basis of discharges (Q) recommended for the key dry-season (Sep) and wet-season (Mar) months (ref. Section on pilot assessment). It was decided that the appropriate % assurance for the maintenance IFR was 70%. This meant that the recommended discharges for each month should be equalled or exceeded 70% of the time on the corresponding one-month daily flow duration curve (FDC) for the river. The figure of 70% was derived from a comparison examination of hydrological data for the Crocodile River, identified by specialists as non-flashy perennial river, with data for rivers of different hydrological regimes (see Figure A1.1).

**B Class maintenance IFR**

| Q | $d_{max}$ | % assurance |
|---|---|---|
| 3.5 $m^3s^{-1}$ March | 0.70 m | 70% |
| 1.5 $m^3s^{-1}$ September | 0.55 m | 70% |

The drought IFR flow and assurance remain the same for all classes and for C1 and C2 scenarios. The duration between drought and maintenance is greater for C1 than C2, but maintenance flows are lower for C2. Long-term average volumes of flow remain the same for C1 and C2.

The maintenance IFR for a B Class Crocodile River was adjusted to represent a C Class River. The decision was taken to determine the flows for only the next lowest full EMC i.e. C Class. The C1 and C2 methods were applied as potential approaches by which the IFR adjustment could be calculated for the river in a C Class condition.

**Procedure for application of C1 method (less assurance)**

Initially, specialists decided that the shift from a B to C class could be addressed simply by a shift in % assurance of the B class maintenance IFR from 70% to 50%. Theoretically, a different % assurance could be set for each month. However, in this test application (and most instances to date) a single % assurance was set for maintenance for all months. Figure A1.2 illustrates the concept although the reduced % assurance value for the 'less assurance' method is different.

For both FDCs representing Class B and Class C, the March maintenance flow remains at 3.5 $m^3$ $s^{-1}$, but the percentage of time for which this discharge is equalled or exceeded is reduced to 50% for Class C. Similarly for Class C, the maximum baseflow is reduced and the time at the drought IFR and between drought and maintenance flows increases. With the C Class curve, there is a higher risk of not meeting the maintenance IFR during drier years. Drought flows are set at 100% assurance (i.e. flows are not permitted to go lower), but in the B Class FDC, drought flows are expected to occur less frequently than in a C class situation.

However, the long term IFR volume with a C class using 50% assurance of maintenance was not significantly different from the same volume of the B class and was much higher than the long term IFR volumes generated using method C2 ('less depth'), following the guidelines based on extrapolation from previous IFR results (Figure A1.3).

Due to the perceived need to increase the difference in volumes between a B Class and a C Class scenario, and to be consistent with the hydrological extrapolation guidelines, the assurance of maintenance flows had to be reduced to less than 50 %. In fact, to achieve the necessary long term volume, the assurance of maintenance flows was reduced to 5% (see Table A1.1).

The two scenarios ('less depth' vs. 'less assurance') were then compared by converting the flows representing the 5%, 50% and 70% assurances to hydraulic parameters, i.e. depths, wetted perimeter and velocity. These depths were checked on the river profiles and ecological reasoning was provided indicating a preference of the C2 scenario. Participants felt that 5% might be too low an assurance, as they perceived that a higher percentage would be acceptable.

**Table A1.1** Comparison of the C1 and C2 Class hydraulic data and discussion of ecological implications

**Crocodile River : PIFR Site 1**

EMC : B
Maintenance flows - March, 3.5 $m^3$ $s^{-1}$

Assurance of maintenance flows : 70%
September, 1.5 $m^3$ $s^{-1}$

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | *DECREASED ASSURANCES (C1)* | | *DECREASED DEPTH (C2)* | |
| | Discharge ($m^3 s^{-1}$) | Hydraulic parameters | Discharge ($m^3 s^{-1}$) | Hydraulic parameters |
| March 5% | 3.5 | 0.7m (Depth)<br>27m (Perimeter)<br>0.35 m/s (Velocity) | 2.02 | 0.6<br>27<br>0.26 |
| March 50% | 1.8 | 0.58 | 1.99 | 0.59<br>27<br>0.26 |
| March 70% | 1.1 | 0.51<br>24<br>0.2 | 1.84 | 0.58<br>27<br>0.25 |
| Sept 5% | 1.5 | 0.55<br>25<br>0.22 | 0.9 | 0.49<br>23<br>0.18 |
| Sept 50% | 0.9 | 0.48 | 0.89 | 0.49<br>23<br>0.18 |
| Sept 70% | 0.6 | 0.43<br>23<br>0.14 | 0.82 | 0.48<br>23<br>0.16 |

Using the cross-sections and associated hydraulic relationships, the hydraulics corresponding with the C Class discharges at 70% assurance were calculated and compared with the B Class hydraulics (Table A1.*1*).

**Ecological implications of the C1 scenario**

In the C1 scenario, the C1 curve indicates that the B Class maintenance IFR has only a 5% assurance in March and September. The C1 discharge and associated hydraulic variables, corresponding with 70% assurance, are given in Table A1.1. Examination of the ecological implications of the reduction in discharge and associated hydraulic variables (from B class), and increase in time for which such conditions would apply (70% of the time) for the C1 scenario, pertain to the effects on a riffle area as follows:

- For a substantial portion of time, even during the dry season, and for both fish and invertebrates the following will be experienced:
  - Large decrease in the proportion of riffle habitat of a wide range of hydraulic variability.
  - Lower quality riffle habitat. Increase in the proportion of zero-velocity areas.

  However, there would still be a range of biotopes available for all species, including riffle-specific species.
- The general trajectory over the long-term would be a degradation of habitat and biotic diversity, resulting in an increased risk of loss of both instream and riparian species.

- With this overall regime, there would be an increase in the degree of stress placed on the system, with more frequent stressful effects and a greater opportunity for ecological damage with additional impacts.
- Increased dependence on tributaries as refuges which increases risk. If the mainstream is stressed under drought conditions, is this problematic with regards to the tributaries, e.g. an additional impact such as a chemical spill then could result in major degradation.
- Increase in biotope patchiness (fragmentation). Loss of range of variability in each biotype.
- General loss of resilience and ability to recover from disturbance. Risk with respect to degree of resilience. Slower recovery (if any) from stressed situation.
- From a geomorphological perspective, in March, an average velocity of 0.2 m $s^{-1}$ is at the lower threshold for sediment entrainment at baseflows (including fines). Therefore, there is expected to be a gradual increase in sediment deposition and hence, increased embeddedness. This would result in a gradual loss of microhabitats for invertebrates (fish diet items).
- For riparian vegetation, in March, the lower terrace, which is inhabited by mesic grasses/reeds, would be stranded in the short-term. Therefore, in the long-term there will be a decrease in this habitat type. Therefore, there will be less habitat for amphibians, invertebrates and fish.
- For riparian vegetation, in September, under baseflow conditions, a narrow marginal zone is required. Participants were uncertain as to whether the decrease in wetted-perimeter would eliminate this wetted area altogether or render it too narrow to be of use. If the decrease in discharge resulted in a sharp decline in the relevant part of the wetted-perimeter curve, this would be a problem.
- For water quality, the situation would only be problematic if flow decreased below 0.6 $m^3$ $s^{-1}$ in the hot months.

Other related issues:

- Difficult to assess floods in this method as dealing with overall flow volumes.
- Degree of natural resistance of the system

Ecological implications pertaining to higher flows:
Higher flows are less frequent under a C Class C1 scenario, but there are no clear indications of the potential implications for the riverine ecosystem. The following are some possible results:

- Increased encroachment of vegetation.
- Decrease in fish and invertebrate species/cues.
- Less bank recharge for vegetation.
- Less frequent channel maintenance and flushing events.
- Less frequent flushing for improvement of water quality.
- Decreased deposition of silt for riparian vegetation seedling establishment, which could result in changes in community structure.
- If sediment deposition increases there is the possible expansion of instream vegetated islands and a possible resultant change in channel type.
- If reed invasion of islands occurred there could be a resultant narrowing of the channel.
- Increased risk of very low flows in the hot summer period. This would probably result in increased temperatures, decreased dissolved oxygen (DO), and increased algae and

associated nutrients. Such changes in water quality would likely result in a loss of suitable benthic habitat for fish and invertebrates.

**Ecological implications of the C2 scenario as compared with the C1 scenario**

The discharges for each of the series of % assurances (5%, 50% and 70%) were calculated from the modified FDC derived from hydrological extrapolation for a C Class Crocodile River. The corresponding hydraulic data were then determined from the PIFR cross-sections (Table A1.1).

Each set of hydraulic information was then evaluated in terms of the ecological changes it represented from a B to a C class, and the results for the C1 and C2 methods were compared.

Points raised from an ecological perspective in comparison of the C1 and C2 methods, as follows:

- The C2 method resulted in more frequent, slightly higher flows than C1. There was thus a marked increase in flow constancy as a result of applying the C2 method. There was a lower amplitude of flows in the C2 scenario.
- Application of both the C1 and C2 methods resulted in regimes that are both much less variable than the natural regime. The C1 method provided a regime that more closely matched the natural regime in terms of baseflow variability than C2. The CV for C1 was 0.6 compared to 0.4 for C2.
- For both approaches, there would be a progressive deterioration in the riverine ecosystem from B to C Class.
- For C2, in March at 70% assurance, there is an increase in wetted perimeter (by 3m) compared with C1. This greater wetted perimeter under C2 would provide additional instream habitat.
- In March, at the 70% assurance level, dry years are closer to drought for C1 than C2 scenario (1.14 $m^3 s^{-1}$ compared with 1.84 $m^3 s^{-1}$).
- Under the C2 scenario, at the 70% assurance level, the river would be deeper and wider for much of the time.
- With degradation from B to C Class, for both C1 and C2 scenarios, there would most likely be a loss of 2 fish species in September (dry season): *Chiloglanis bifurcus* needs sufficient depth and velocity over riffles. Thus, the scenarios reflected by C1 and C2 would probably ultimately result in degradation beyond C Class to a D Class river.
- If there would be a loss of flow variability due to an anthropogenic impact in the tributaries, the C2 scenario would probably become less desirable than the C1 scenario.

In comparing the two scenarios it was clear that there would need to be a trade-off between increased constant, available habitat and decreased flow variability. It was generally agreed that for a Class C river it was preferable to opt for an increase in habitat, with acceptance of the loss of flow variability. It was noted that for a Class B river, neither option would be suitable as for example, the velocities associated with the IFRs were very low.

**The following suggestions and comments on an appropriate method were made**

- Have separate paths for selection of either the C1 or C2 method, depending on the degree of flow variability exhibited by the study river. According to the hydrologist, this is not feasible and too complex.

- Generate a combination C1 - C2 type curve.

  Apply both methods at the level of the Desktop Estimate. However, this was recognised as a problem if the two methods result in discrepancies in the recommended % MARs. It was agreed that it would not be appropriate to apply both methods to calculate Desktop Estimates as differences in MARs would likely reduce the confidence level of the estimate.

- There needs to be feedback into the DSS.

**Summary and conclusions**

In summary, the C1 method resulted in higher maximum baseflows, but lower flows at the 50 - 70% assurance range. In contrast, the C2 method resulted in very regular inter-annual flow patterns.

The specialists concluded that both options would lead to progressive degradation of the Crocodile River from the recommended Class B status (with its associated IFR recommendation) into Class C (and possibly beyond). However it was felt that the C2 approach was marginally less detrimental because, at frequently occurring baseflows, the suite of hydraulic variables (maximum depth ($d_{max}$), average velocity ($\nabla$, and wetted perimeter (w.p.)) indicated that more extensive habitats were available for fish, invertebrates and riparian vegetation.

The main problem with the C2 method is that it is difficult to apply within the Intermediate Determination, because specialists have great difficulty in defining and motivating for specific flows for a range of EMCs outside the current EMC represented by the river in its present state. It was really only possible to list the ecological consequence of having a C Class river defined by a flow regime calculated using the C2 method. In addition, the C1 method appears to fit more conveniently within a risk-based process. The output of a series of flows corresponding with different percentage assurances, according to EMC, is directly comparable with the percentage assurances assigned to different offstream uses of water under different management scenarios.

**APPENDIX 2**
**IFR DATA TO BE USED IN THE EVALUATION OF 'LESS FREQUENCY/ASSURANCE' VS. 'LESS DEPTH' METHODS**

**MKOMAZI RIVER : IFR SITE 2**

EMC : B
Maintenance flows - March, 10 $m^3/s$

Assurance of maintenance flows : 60%
September, 2.3 m3/s

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | DECREASED ASSURANCES (C1) | | DECREASED DEPTH (C2) | |
| | Flow rate ($m^3/s$) | Hydraulic parameters | Flow rate ($m^3/s$) | Hydraulic parameters |
| March 10% | 10 | 0.95 m (Depth)<br>48.4 m (Perimeter)<br>0.51 m/s (Velocity) | 7.82 | 0.88<br>45.4<br>0.47 |
| March 60% | 6.48 | 0.83<br>43.3<br>0.45 | 7.23 | 0.86<br>44.5<br>0.47 |
| September 10% | 2.3 | 0.6<br>27.7<br>0.36 | 1.7 | 0.55<br>24.7<br>0.34 |
| September 60% | 1.23 | 0.49<br>21.6<br>0.32 | 1.54 | 0.53<br>23<br>0.34 |

**MKOMAZI RIVER : IFR SITE 4**

EMC : B
Maintenance flows - March, 12.5 $m^3/s$

Assurance of maintenance flows : 60%
September - 3.5 $m^3/s$

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | DECREASED ASSURANCES (C1) | | DECREASED DEPTH (C2) | |
| | Flow rate ($m^3/s$) | Hydraulic parameters | Flow rate ($m^3/s$) | Hydraulic parameters |
| March 10% | 12.5 | 1.26 m (Depth)<br>88 m (Perimeter)<br>0.34 m/s (Velocity) | 10.32 | 1.21<br>85.4<br>0.32 |
| March 60% | 7.8 | 1.14<br>75<br>0.3 | 8.82 | 1.17<br>82.4<br>0.31 |
| September 10% | 3.5 | 0.95<br>46.6<br>0.22 | 2.7 | 0.9<br>45.1<br>0.2 |
| September 60% | 1.94 | 0.84<br>41.6<br>0.12 | 2.4 | 0.88<br>44<br>0.2 |

**THUKELA RIVER : IFR SITE 0 (SKIETDRIFT)**

EMC : B
Maintenance flows : February - 10 $m^3/s$

Assurance of maintenance flows : 60%
August - 2.5 $m^3/s$

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | DECREASED ASSURANCES (C1) | | DECREASED DEPTH (C2) | |
| | Flow rate ($m^3/s$) | Hydraulic parameters | Flow rate ($m^3/s$) | Hydraulic parameters |
| February 10% | 10 | 0.52 m (Depth)<br>47.1 m (Perimeter)<br>0.55 m/s (Velocity) | 6.96 | 0.43<br>46.1<br>0.49 |
| February 60% | 4.77 | 0.36<br>45.3<br>0.44 | 5.96 | 0.40<br>45.7<br>0.47 |
| August 10% | 2.5 | 0.26<br>42.7<br>0.39 | 1.91 | 0.23<br>41.9<br>0.38 |
| August 60% | 1.43 | 0.20<br>41.0<br>0.38 | 1.75 | 0.22<br>41.6<br>0.38 |

**SAND RIVER : IFR SITE 8 (KNP)**

EMC : B
Maintenance flows : March - 2 $m^3/s$

Assurance of maintenance flows : 70%
September - 0.47 $m^3/s$

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | DECREASED ASSURANCES (C1) | | DECREASED DEPTH (C2) | |
| | Flow rate ($m^3/s$) | Hydraulic parameters | Flow rate ($m^3/s$) | Hydraulic parameters |
| March 10% | 2 | 0.79 m (Depth)<br>22.4 m (Perimeter)<br>0.28 m/s (Velocity) | 1.45 | 0.72<br>21.3<br>0.26 |
| March 70% | 0.85 | 0.61<br>17.3<br>0.24 | 1.24 | 0.68<br>20.0<br>0.26 |
| September 10% | 0.47 | 0.51<br>13.7<br>0.23 | 0.36 | 0.47<br>11.7<br>0.23 |
| September 70% | 0.19 | 0.39<br>12.5<br>0.23 | 0.29 | 0.44<br>9.7<br>0.23 |

**MHLATHUZE RIVER : IFR SITE 3**

EMC : B/C
Maintenance flows : February - 3 $m^3/s$

Assurance of maintenance flows : 60%
August - 0.9$m^3/s$

| ASSURANCES (maintenance) | SCENARIOS | | | |
|---|---|---|---|---|
| | DECREASED ASSURANCES (C1) | | DECREASED DEPTH (C2) | |
| | Flow rate ($m^3/s$) | Hydraulic parameters | Flow rate ($m^3/s$) | Hydraulic parameters |
| February 10% | 3 | 0.60 m (Depth) | 2.12 | 0.53 |
| February 60% | 1.3 | 0.45 | 1.77 | 0.5 |
| August 10% | 0.9 | 0.4 | 0.69 | 0.37 |
| August 60% | 0.49 | 0.33 | 0.61 | 0.35 |

## APPENDIX 3
## RESULTS OF SCENARIO COMPARISONS

The IFR data in Appendix 2 was used to evaluate the usefulness of the two methods (i.e. the 'less frequency/assurance' method vs. the 'less depth' method) for determining maintenance flows for different EMCs. A group of workshop participants was allocated one or two sites on which to test the two methods or scenarios. Their conclusions are listed below; see Appendix 2 for raw data. The IFR workshop documents and site photographs were also available for use. The Crocodile River, PIFR Site 1 (Appendix 1, Table A1.1), was not discussed as the data was analysed in the workshop starter document (Appendix 1).

### THUKELA RIVER: IFR SITE 0 (SKIETDRIFT)

*O'Keeffe, Louw, Scherman*

The site selected is a fairly flat cross-section over rocky bedrock, with steep banks. Reductions in flow would therefore reduce depth and velocities, but not wetted perimeter (not until very low flows are reached). **The 'less assurance' option results in rare base flows, higher than the 'less depth' option, but lower more frequent flows.**

The approach taken in this exercise was to try and assess composite hydraulic habitat differences between the two options, and to express this as a difference in risk to the biota.

Theory: Habitat is to be used as a surrogate for biota, and assume that losses in depth, velocity and perimeter all represent inter-dependent but additive stresses to the biota.

*Example 1*

Comparing the February flows for each option at the 10% assurance level (see Appendix 2):

'Less assurance' option (10 $m^3/s$) provides more habitat than the 'less depth' scenario (6.96 $m^3/s$):

20% more depth
2% more perimeter
10% higher average velocity

Using the risk assessment calculation method (Jooste, pers. comm.)

D + P + V- (D.P + D.V + P.V) + ? (minimal expression)
0.2 + 0.02 + 0.1 - (0.004 + 0.02 + 0.002 ) + ?
= 0.29
i.e. 29% greater risk of stress.

*Example 2*

Comparing the August flows for each option at the 60% assurance level (see Appendix 2):

'Less depth' option (1.75 $m^3/s$) provides more habitat than the 'less assurance' method (1.43 $m^3/s$):

10% more depth

3% more perimeter
no difference in average velocities

Using the formula as in Example 1:
0.1 + 0.03 - (0.003) + ?
~ 0.13
i.e. 13% greater risk of stress.

However, the relationship between stress and flow is not linear (see Figure A3.1). The usual assumption is that initial reductions in flow (from natural levels) cause very little stress, but stress increases exponentially as flow reductions exceed ± 70 - 80%.

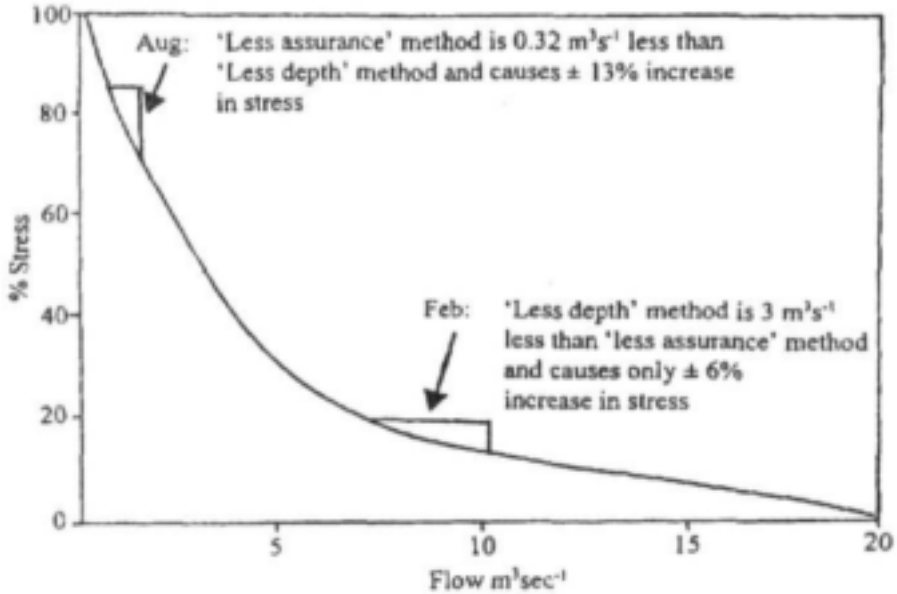



**Figure A3.1** Assumed relationship between flow and instream biotic stress. Superimposed is the relative increase in risk of stress of 'less frequency' and 'less depth' February flows at 10% assurance.

N.B. In reality, separate stress curves would need to be generated for each different month.

**Conclusion**

Since we do not at present have accurate estimates of stress curves, it is very difficult to compare the stress risks of the 'less assurance' and 'less depth' options. The following are therefore the best assessments of the 2 options:

- In the long term there will be very little difference in the changes in the river caused by either option, since we assume that the high flow scenarios will be the same using both methods. These are flows which govern sediment transport and therefore channel morphology. Base flows under both scenarios are inadequate to move sediment in the channel.

- The 'less assurance' method provides rare higher base flows than the 'less depth' method and more flow variability, with less loss of the natural disturbance regime.
- The 'less depth' method provides more constant and higher low flows, and therefore more refuge habitat during stress periods than the 'less assurance' method.
- 'Less assurance' flows have more consistent ecological motivation than the 'less depth' scenario as the recommended flows stay the same, but with a decreased frequency, while 'less depth' flows would require separate ecological motivation.
- On balance, the 'less assurance' method seems slightly more appropriate for this site.

**MHLATHUZE RIVER: IFR SITE 3**
*O'Keeffe, Louw, Scherman*

This is a wide sandy bed site, very flat, but with deeper channels at each side, and fringed with reeds. The important aspect of this site (which is still largely natural) is to maintain these side channels as reasonably deep habitats for fish (especially for passage up and down the river) and as marginal vegetation habitats for invertebrates and fish. There was limited hydraulic information available, since only depths were provided, and no composite index of hydraulic habitat was possible.

In essence, the same differences as for Site O on the Thukela River apply, except that the shallow habitats in the centre of the river disappear rapidly with flow reductions, and the only effective habitat remaining is the margins.

The same conclusions apply, i.e. rare higher flows under the 'less assurance' scenario, but more consistent higher low flows under 'less depth'. In this case the importance of fish passage is better catered for by occasional deeper marginal habitats than by constantly higher low flows, and therefore the 'less assurance' option is once again slightly to be preferred.

**Conclusion**

In the two sites considered, the 'less assurance' option was marginally better from an ecological perspective. Since it is also preferable from the point of view of river management (the % assurance method is applied to user requirements), and because the motivations remain consistent, we conclude that it should normally be the option of choice. It seems possible that the 'less assurance' option is more appropriate for variable ('flashy') rivers, and that the 'less depth' option could be considered for more constantly flowing rivers, in which the biota may be less resilient to wide fluctuations in flow.

**MKOMAZI RIVER: IFR SITES 2 AND 4**

*Rowntree, Kemper, Jooste*

| SITE | PERIOD | 'LESS ASSURANCE' METHOD | 'LESS DEPTH' METHOD |
|---|---|---|---|
| Mkomazi IFR 2 | wet | MOTIVATION: water depth and velocity over cobble bar to maintain fish habitat, basal area of *Arundinella* grass inundated | |
| | | significant stress with high risk for shorter periods, loss of cobble bar habitat, loss of critical habitat. Velocities may be significantly reduced in terms of filter feeding (invertebrates) and sedimentation; higher short term flows would assist winnowing of sediments; variability maintained. | continuous moderate stress, reduced habitat quality, increased water temperatures over cobble bar, reduced velocities and increased siltation; reduced variability. |
| | dry | MOTIVATION: flow maintained in deeper channels, water depth just wetting roots of *Arundinella* grass. | |
| | | significant stress over extended periods, possible change to vegetation structure, increased variability | continuous moderate stress but may be acceptable, reduced variability |
| Mkomazi IFR 4 | wet | MOTIVATION: Basal wetting of riparian vegetation; shelter in secondary channels for small fish plus deep water habitat for adults; aquatic invertebrate habitat | |
| | | temporary loss of nursery areas for fish, increased variability | permanent loss of satisfactory nursery areas for fish |
| | dry | MOTIVATION: wet base of island on right-hand side in order to wet base of reeds, provision of habitat for various species | |
| | | reduced aerial extent of habitat but little change in quality, variability maintained | slight reduction in aerial extent of habitat, little variability. |
| KEY | | Preferred option | Preferred option but uncertainty high |

General: If reduced flows (i.e. 'less depth') under reduced assurance fall below a critical threshold that needs to be maintained permanently, then reduced depth is probably the better option, as long as threshold depths are maintained. Otherwise reduced assurance is probably the better option as variability is maintained. Important questions are: what are the critical habitats, for how long do they need to be maintained and how is their availability affected by the different flow regimes?

**SAND RIVER: IFR SITE 8 (KNP)**
*Tharme, Birkhead, Kleynhans, Thirion*

*Note: Appendix 2, Sand River, IFR Site 8 (KNP) Table: September 70%; for 'decreased assurance' scenario read Q= 0.19 $m^3 s^{-1}$, wetted perimeter is 6.4 m (not 12.5).*

**General process adopted**

The results of the application of each method were evaluated independently and then compared with each other for IFR 8, classed as a braided sand-bed reach of the Sand River. For each method, the March and September results were assessed separately. The following information was used for this purpose. All available photographs of IFR 8 at different discharges were consulted. The ecological and geomorphological motivations pertinent for the specific base flows represented by the two scenarios, as documented at the Sabie-Sand IFR workshop, were consulted (Tharme, 1997). The cross-section data for IFR 8 were used to assess the ecological implications of the various changes in depth and associated hydraulic conditions across the channel, for the two scenarios. Note that the maximum depths pertained to the deepest channel, a secondary channel, rather than to the active, main channel.

**Decreased assurance scenario**

***March (wet season)***
The reduction in % assurance for the IFR base-flow recommendation for a shift from Class B (as recommended in the IFR workshop) to Class C has the following main implications:
- The maintenance IFR of 2.0 $m^3 s^{-1}$ would occur only 10% of the time.
- A 58% lower discharge of 0.85 $m^3 s^{-1}$ would occur 70% of the time (i.e. 60% more of the time than under a Class B scenario).

Comparisons of the hydraulic data corresponding to the March 10% and 70% base flows (Table in Appendix 2) showed a decrease of 23% for maximum depth ($d_{max}$), 23% for wetted perimeter (w-p); and 14% for average velocity ($v_{av}$). However, note that the $d_{max}$ for the active channel is *ca* 0.46 m at 2.0 $m^3 s^{-1}$ and *ca.* 0.28 m at 0.85 $m^3 s^{-1}$, i.e. a reduction of 39% (as the depth data in Appendix 2, pertains to the deepest channel at the site, which is not part of the main, active channel).

The following ecological effects would probably be associated with this scenario:
- A decrease in wetted marginal vegetation, primarily reeds (*Phragmites mauritianus*). However, there would still be some inundation of reeds along the margins of the active channel, due to the relatively even cross-sectional bed profile at the site (at 0.5 $m^3 s^{-1}$ some reeds would still be inundated). In addition, reed growth is highly dynamic and reeds would likely encroach into the channel in the long-term; concomitant with a decrease in discharge.
- Minnows would still have some access to marginal reeds for cover.
- A loss of about 0.2 m in depth. The dominant biotopes are runs, with few pools. As the channel profile is fairly uniform, much of the cross-section would be at about the same depth of 0.28 m, resulting in a major loss of deeper-water biotopes for fish. A maximum

depth of 0.28 m is considered marginal for the survival of *Chiloglanis swierstrai*. Hence, this species might be lost from the river, especially if there are synergistic stresses.

- The homogenous substratum of sand, with very few bedrock outcrops, provides relatively poor habitat for invertebrates. Additional species would be reliant on marginal vegetation. Although there would likely be a decrease in species abundances with the reduction in available wetted habitat, there would still be adequate areas of instream and marginal habitat. Loss of species diversity would probably be minimal. Depth would probably be less important than velocity in the provision of instream sand-bed habitat. As the velocity distribution would be fairly homogenous across the channel, there probably would be adequate invertebrate habitat.
- The decrease in discharge would not result in a major loss of longitudinal or lateral connectivity of the channel. Although some braid bars would become isolated, the primary, active channel would remain connected. The reduction in the cross-sectional area available for fish passage would not be limiting.
- From a geomorphological perspective, the decrease in discharge would still allow some sediment transport, as measurable transport has been observed at only 0.5 $m^3 s^{-1}$. In addition, there is still some flow variability under this scenario. However, the decrease in discharge for more of the time would mean that effectively less sediment would be moved through the system, so there is a possibility of aggradation. This would be dependent on the high flow regime. Reduced sediment transport would possibly result in filling of the eddy scour holes around bedrock/boulders, which are known to be important habitat for many fish species.

***September (dry season)***

The reduction in % assurance has the following main implications:

- The maintenance IFR of 0.47 $m^3 s^{-1}$ would occur only 10% of the time.
- A 60% lower discharge of 0.19 $m^3 s^{-1}$ would occur 70% of the time.

A comparison of the hydraulic data corresponding to the September 10% and 70% base flows (Table in Appendix 2) showed a decrease of 24% for $d_{max}$; 53% for w-p; and negligible change for $v_{av}$.

The following ecological effects would probably be associated with this scenario:

- Although velocities are similar under both flows, the marked decrease in depth with the 70% assurance discharge would result in very shallow flowing water across only 53% of the cross-section. As it is the dry season, this would probably result in decreased water quality, particularly temperature and dissolved oxygen. The decrease in these variables would probably seriously detrimentally affect both fish and invertebrates. The shallower waters would increase the vulnerability of fish to predation.
- Geomorphologically, there would probably be no major problems with flow reduction. It was noted that it might represent an improvement over the B Class recommended discharge. It was noted that the Sand River is regarded by several scientists as a seasonal river historically, ceasing to flow fairly often in this month.
- As there are only two main biotopes at the site i.e. runs over sand and marginal vegetation, the decreased flows would not result in major habitat loss in terms of substratum types and velocity. However, there would be critical loss of depth and sand bed habitat with fringing marginal vegetation.

- It was noted that there probably would not be any flood events in September to ameliorate the situation of low base flows, such as flushing out poor water quality or redistributing any accumulated sediments.

**Lower depth scenario**

***March (wet season)***
The reduction in depth for the IFR base flow recommendation for a shift from Class B 70% assurance (as recommended in the IFR workshop) to Class C has the following main implications:

- The maintenance IFR would decrease by 38% from 2.0 $m^3 s^{-1}$ to 1.24 $m^3 s^{-1}$, for 70% of the time. The curve is flatter, so there is less difference between the discharges corresponding with a 10% and 70% assurance. In terms of maximum discharge, there is a reduction from a B Class maximum of about 2.4 $m^3 s^{-1}$ to 1.45 $m^3 s^{-1}$. Comparisons of the hydraulic data (Table in Appendix 2) showed a decrease of 24% for $d_{max}$ in the main channel (as opposed to data for the deeper side channel), representing a decrease of 0.11m from a B Class depth of *ca.* 0.46m to 0.35m, 11% for w-p, and 7% for $v_{av}$.

The following ecological effects would probably be associated with this scenario:

- Geomorphologically, there would be less sediment transport, over a smaller range of discharges. This would be undesirable for a sand bed river.
- For fish and invertebrates, water depths would still be acceptable, especially as the cross-section profile is fairly uniform (average depth would approximate $d_{max}$).
- As there is little change in velocities, there would not be a problem.
- Marginal vegetation is likely to remain inundated as there is only a small (*ca.* 1m) loss of wetted perimeter on each side (even cross-section profile). There possibly would be some short-term loss of invertebrate production. The assumption was made that *Phragmites mauritianus* would respond rapidly in the long-term.

***September (dry season)***
The reduction in depth for the IFR base flow recommendation for a shift from Class B 70% assurance (as recommended in the IFR workshop) to Class C has the following main implications:

- The maintenance IFR would decrease by 38% from 0.47 $m^3 s^{-1}$ to 0.29 $m^3 s^{-1}$, for 70% of the time. The curve is flatter, so there is less difference between the discharges corresponding with a 10% and 70% assurance. Comparisons of the hydraulic data (Table in Appendix 2) showed a decrease of 39% for $d_{max}$ in the main active channel (as opposed to data for the deeper side channel), representing a decrease from *ca.* 0.18m to 0.11m; 29% for w-p; and a negligible difference for $v_{av}$.

The following ecological effects would probably would be associated with this scenario:

- Velocity would not be an issue, but maximum depth would decrease markedly in the active channel. This would be a serious problem for maintaining a viable adequate population of *Chiloglanis swierstrai*.
- Large scale loss of depth would likely reduce the suitability of available habitat for invertebrates.
- There would be a sediment transport problem, with little transport taking place at such low discharges. Without adequate sediment transport, there would possibly be infilling of

localised, eddy scour pools around bedrock/boulders. These areas are known to be important habitat for many fish species.

- There would be a small loss of wetted habitat for invertebrate production.

**Comparison of the two scenarios derived from application of the 'lower assurance' and 'lower depth' methods**

The following probable ecological implications pertain to the two C Class scenarios:
Both scenarios would result in a degradation in ecological condition of the river over time, into a C class and possibly lower.

In general ecological terms, the 'lower depth' scenario would represent the establishment of a smaller river with slightly less habitat. This would be in contrast to the 'lower assurance' scenario where the river would be larger, but with less water in it for some of the time.

Although both scenarios would represent an overall reduction in flow variability, the 'lower assurance' scenario would retain more variability than the 'lower depth' one. The latter scenario would result in a more constant base flow regime, with increased constancy of available habitat.

From a geomorphological perspective, the 'lower assurance' scenario is considered preferable as it provides improved flow variability to enable sediment transport over a wider cross-sectional area of the channel. This would assist in preventing active channel incision in the sand bed. Under the 'lower depth' scenario, there would be less sediment transport over a smaller range of discharges (March). This would contribute to the development of an incised smaller channel within the sand bed i.e. similar morphology characteristic of the Letaba River.

For fish, the 'lower depth' scenario would be preferred. Although it resulted in a greater decline in depths than the other scenario during the wet months, there would likely be flood events at that time. Importantly, it provided more suitable depths during the dry season than the 'lower assurance' scenario, where there was an extreme reduction in depth at the time of lowest flow in the river. It was noted that neither scenario would provide adequate depths for continued survival of a population of *Chiloglanis swierstrai*. Moreover, the scenario is preferred provided that there is inundation of marginal vegetation, which would still occur (because of the dynamic nature of reed growth, uniformly flat cross-section profile and fairly wide riparian fringe).

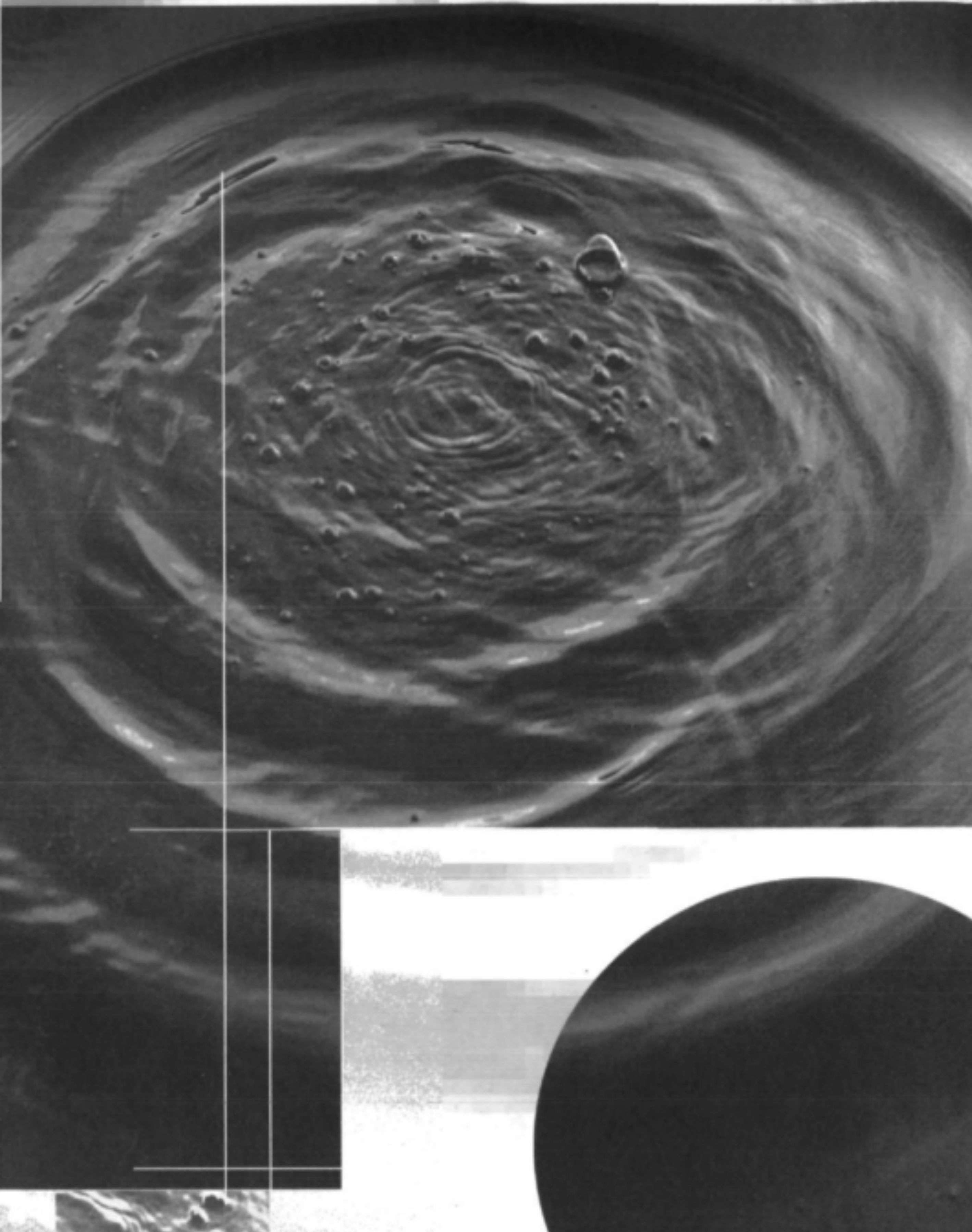
For invertebrates, the 'lower depth' scenario would appear more suitable due to reduced loss of wetted perimeter and hence available habitat (29% cf. 53% for September at the 70% assurance level). If the river has pest invertebrate species, however, the 'lower assurance' scenario would ensure more flow variability and reduce flow constancy, relative to the alternative scenario.

*Note: A member of the team voiced concerns regarding the principle of lowering a desired Ecological Management Class. If required, it was felt that a much more detailed analysis of the (perceived) ecological consequences would be necessary, and that more effort should be put into typing the river ecologically (to determine if it would be sensitive to certain changes, or if responses would merely be an imitation of natural disturbances). It is therefore necessary to determine how sensitive the selected river is toward various kinds of disturbances or modifications. A decision support system might be useful in this regard.*

**Reference**

Jooste, S. Personal communication. IWQS, DWAF, Pretoria.

Tharme, R.E. (1997). Sabie-Sand River system instream flow requirements: proceedings of IFR workshop. Aan de Vliet, 18-22 August 1996. Southern Waters Report. September 1997. Commissioned by Department of Water Affairs and Forestry and Ninham Shand Consulting Engineers.



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