EVALUATION OF THE ROLE OF SASS4, AS AN AQUATIC BIOMONITORING METHOD, IN THE ECOLOGICAL RISK ASSESSMENT PROCESS AND IN THE DETERMINATION OF RESOURCE DIRECTED MEASURES FOR THE LUVUVHU RIVER

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ABSTRACT

The focus of the National Water Act (Act 36 of 1998) is on the sustainable utilization of our water resources. This is to be achieved through the implementation of an integrated resource protection approach, which is aimed at ensuring that a balance is maintained between the protection and utilization of our countries water resources. This approach sets Resource Quality Objectives (RQOs) that define acceptable levels of water resource protection. The acceptable risk of damage to the ecological integrity of a water resource will play an important role in the setting of these objectives, e.g. for a water resource of lower importance a higher risk would be acceptable with the subsequent setting of RQOs at less stringent levels.

A desktop Resource Directed Measures (RDM) determination has already been performed for the Luvuvhu River in the Northern Province. There was decided to evaluate a facet of this desktop study, namely the Present Ecological Status (PES), by utilising the South African Scoring System version 4 (SASS4) and the Integrated Habitat Assessment Method (IHAS) biomonitoring techniques. It was then possible to compare the desktop determined PES to the PES determined from the information provided by the biomonitoring techniques. Ultimately, SASS4 verified the reliability of the RDM methodology

Further, to facilitate the introduction of Ecological Risk Assessment (ERA) into South Africa there was looked at how and where SASS4 and IHAS would fit into the various phases of the ERA process. SASS4 serves as an indication of the extent of an impact, and in conjunction with an ERA, would provide the means with which to determine causality. A retrospective ERA based on data obtained from SASS4 and a concomitant habitat assessment method will thus provide a valuable tool for the protection of our water resources.

CHAPTER 1

INTRODUCTION

1.1 DEFINING SOUTH AFRICA'S WATER PROBLEM

Water in South Africa is a scarce commodity, where the influences of climate, topography, and average annual evaporation contribute to our water shortage. Pollution and the inefficient use and management further limit the quality and quantity of our water resources.

South Africa is considered to be a dry country, where the climate ranges from semi-arid to hyper-arid (Davies and Day, 1998). Annual rainfall is approximately 497 mm (Dallas, 1995), and distributed unevenly over the country, where the west is drier than the east. In most areas evaporation also far outstrips precipitation. South Africa is also afflicted periodically by severe and prolonged droughts which are often terminated by severe floods (DWAF, 1986).

The existing water problem is further complicated by the high rate of South Africa's population growth. Davies and Day (1998) illustrated how water supply will, at best, no longer meet demand between 2020 (use of all surface water) and 2040 (use of surface and ground water), and in the worst case (i.e. highest population growth) water will be fully committed between 2003 and 2015 (refer to Figure 1.1).

According to O'Keeffe (1986), river uses in South Africa include: agriculture, urban complexes, population concentrations in rural areas, industry, and recreation, where agriculture accounts for roughly 73% of the total amount of the water used, through mainly water abstraction. These uses have numerous impacts on rivers, including partial or total destruction of the natural river biota, alterations to river functioning, overloading of self-cleansing mechanisms and a concomitant drastic lowering of water quality (O'Keeffe, 1986).

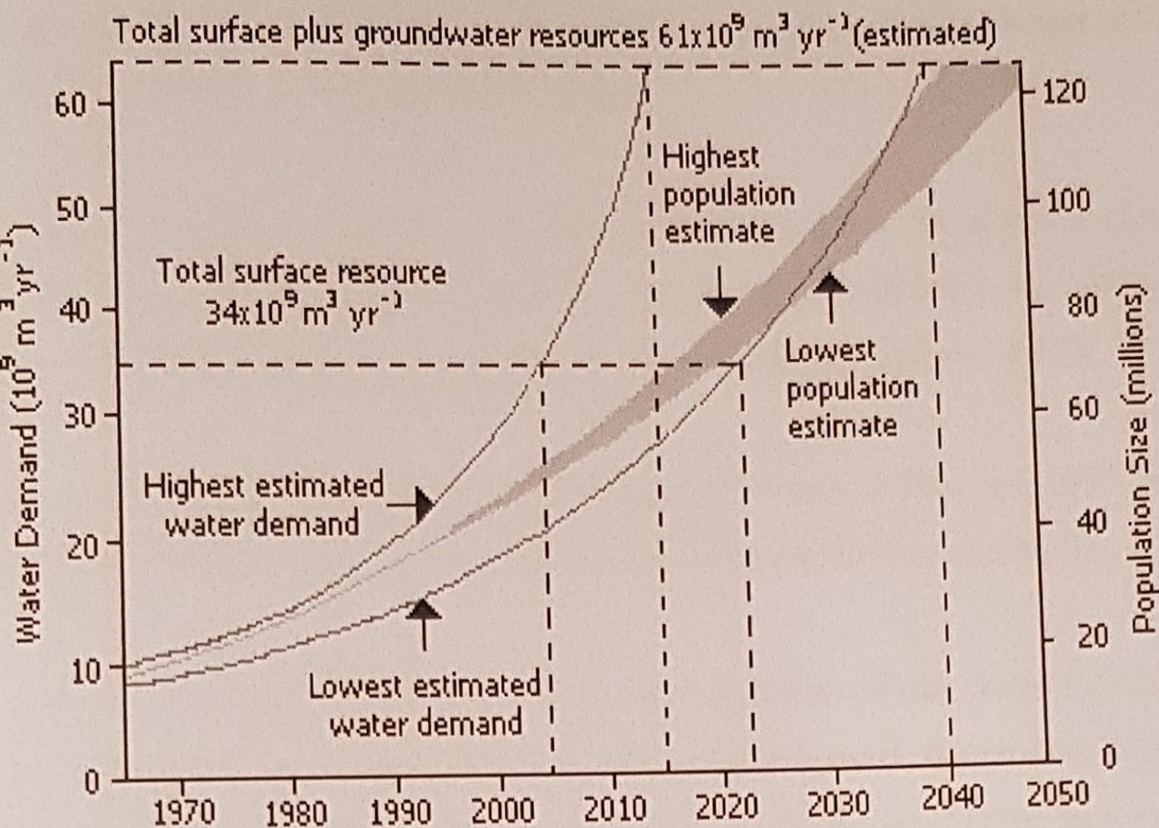

Figure 1.1: The relationship between demand for water and size of the human population of South Africa (Davies and Day, 1998).

Over the past few years, large-scale urbanization of previously rural populations, coupled with growing industrialisation and rapid socio-economic changes, have increased both the demand for water and the extent of impacts on the quality of water resources in South Africa (Roux *et al*, 1997). Many changes are linked to the political reform that our country has undergone, since the move to a democratic government in 1994.

1.2 FINDING A SOLUTION

1.2.1 Adjustment of Legislation

With all these challenges facing our water resources, the Water Act of 1956 was no longer an adequate tool with which to face the future (DWAF, 1986), where water pollution in South Africa had primarily been controlled by applying a uniform effluent standard (Heath, 1993). Further, current water-quality monitoring and management approaches were clearly inadequate to protect the ecological processes that are essential for maintaining the usefulness of the resource (Roux *et al*, 1999). New water policy and legislation was required to improve management of South Africa's water resources.

The ensuing comprehensive water law reform aimed to meet political and social goals of equitable water access, and provided the opportunity to develop an ecologically sound legal and policy basis for water resource management (Palmer, 1999).

The Water Law Principles of 1996, the National Water Policy of 1997 and the National Water Act (Act 36 of 1998) all focused on sustainability and equity.

The purpose of the NWA (Act 36 of 1998), the foundation of our water legislation, is to ensure that our nation's water resources are protected, used, developed, conserved, managed and controlled in ways which take into account amongst other factors-

- Promoting the efficient, sustainable and beneficial use of water in the public interest;
- Protecting aquatic and associated ecosystems and their biological diversity;
- Reducing and preventing pollution and degradation of water resources.

The modification and degradation of any water resource will jeopardise its ability to serve as a sustainable resource. As "renewable natural resources", water resources have a certain amount of resilience to the pressures and demands of utilisation, but if, however, a water resource is over-utilised or allowed to degrade too far, the ecological integrity of the resource can be damaged (DWAF, 1999). According to DWAF (1997), ecological integrity can be defined as 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 characteristics of ecosystems within a specific region. Sustainable development thus endeavors to utilize water resources with a degree of maintenance of the natural character of aquatic ecosystems.

In order to ensure that utilisation of water resources can be sustained in the long term, the structure and function of ecosystems have to be protected (DWAF, 1999). The structure of any ecosystem comprises of biotic (producers, consumers, and decomposers) and abiotic (physical and chemical components) components. O'Keeffe (1986), identified the following functions of a river ecosystem, that are relevant to natural ecosystem processes:

- water supply
- sediment transport
- nutrient transport and recycling
- biotic dispersal
- vegetation maintenance
- water storage
- effluent transport
- flood buffering capacity.

In September 1999, DWAF published a set of documents titled: Resource Directed Measures for Protection of Water Resources (DWAF, 1999). The aim of these documents is to manage and regulate pollution and land use impacts on the water environment, so as to protect water resource quality. This goal is to be achieved through generating Resource Quality Objectives (RQO). A protection-based classification system will provide the framework and context for determination of these objectives as part of Resource Directed Measures (RDM) for water resources. The final outcome is to determine the Reserve (i.e. the water needed to protect basic human and environmental needs) and to manage water uses so as to meet the Reserve (DWAF, 1999). Refer to Chapter 7 for further discussion on this policy.

1.2.2 Assessing the Risk Associated with Water Utilization

Resource quality objectives (RQOs) for a water resource are set on the basis of acceptable risk: that is, the less risk we are prepared to accept of damaging the Resource Base and possibly losing the services provided by the water resource, the more stringent would be the objectives (DWAF, 1999). 'Acceptable risk' is better described by the 'acceptable damage to the ecological integrity', as the acceptable risk to a certain water resource can be regarded as dissimilar between different water users (e.g. recreational vs. industrial users).

As far back as 1986, DWAF already realised that there is an increase in the importance of risk analysis in South Africa. Since then, there has been an increase in the need to study existing risk literature. This will serve to aid in the understanding of the concept of risk and assist in its introduction into our country.

The U.S. Environmental Protection Agency (USEPA) has developed an environmental analysis process, known as ecological risk assessment (ERA). ERA "evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors" (USEPA, 1992). The process's framework and all other relevant information is outlined in the *Guidelines for Ecological Risk Assessment* (USEPA, 1998). In the United States the framework has gained wide acceptance as the basis for developing ecological risk assessment methods and organizing risk assessments within many federal and state agencies (Menzie and Freshman, 1997).

According to Cook *et al.* (1999), ERA begins with a problem formulation phase that defines the contaminant sources, the receiving environment, and the assessment endpoints. Then, for each endpoint, there is an analytical phase consisting of exposure assessment and effects assessment. Finally, the risk characterization phase combines the components of the analysis phase.

Problem Formulation is the most critical step in ecological risk assessment because it provides direction for the analysis and should take into account the ecological, societal, and political issues related to the questions being addressed (Menzie and Freshman, 1997). During the analysis phase, data are evaluated to determine how exposure to stressors is likely to occur (*characterization of exposure)* and, given this exposure, the potential and type of ecological effects that can be expected (*characterization of ecological effects)*. Finally, during risk characterization, the exposure and stressor-response profiles are integrated through the risk estimation process.

In accordance with the Water Resource Protection Policy (DWAF, 1999), risk will soon play an integral part in the protection of our water resources. It has thus been planned to incorporate the USEPA's ecological risk assessment into the NWA (Act 36 of 1998), and apply the concept of risk to the protection of water resources, to determine the potential role that this environmental management tool could play in the management of South Africa's water resources.

Besides a legislative application, an ERA on its own can be conducted to evaluate the potential occurrence of hazards to any natural resources. ERAs hold many similarities to Environmental Impact Assessments (EIAs). EIAs in South Africa are usually conducted according to the Integrated Environmental Management (IEM) procedure, and many of the ecological requirements specified in the IEM procedure are dealt with within the ERA framework (Murray and Claassen, 1999). The problem formulation phase in ERA addresses all the issues required for the scoping phase of the EIA, and the exposure assessment step in ERA covers all the relevant issues, as well as ecological components, specified in the proposed outline of the IEM project proposal (Murray and Claassen, 1999). There is thus a clear-cut resemblance between the ERA and IEM principles and procedures. According to Rosenberg and Resh (1993), future environmental research work should involve much more risk assessment and environmental impact assessment should include prediction and should lead to follow-up work to test those predictions.

1.2.3 Managing Water Resources

Management is the execution of planned controls so as to achieve a desired outcome (Fuggle and Rabie, 1992). According to DWAF (1999), once Resource Quality Objectives (RQOs) have been set for a resource, then those objectives would serve as a basis for water resource management. According to Hugo *et al*. (1997), resource management is a decision-making process in which optimal solutions regarding the manner, timing and allocation of resource use are sought within the economic, political, social and institutional framework. For risk assessment to reach its full potential in our country, it needs to influence this decision-making process.

Suter (1993) identified several advantageous properties of risk assessment in environmental decision making:

- Risks are compared and prioritized on quantitative bases, where possible.
- It provides a systematic means of improving the understanding of risks.
- Risk assessment estimates clear consistent endpoints, in contrast to assessments where unstated and ambiguous endpoints are chosen.
- Risk assessment reduces biased decisions, by separating risk analysis from risk management.

Effective decision-taking and resource management depend, however, on the information provided by effective resource monitoring (DWAF, 1997). Further, legislation supports biomonitoring by defining the environmental objectives and priorities, and provides enforcement actions. Chapter 14 of the NWA (1998) encourages monitoring as a means of protecting and controlling our water resources.

1.2.4 Assessing the Biotic Integrity of the Water Resource

Any utilization of water poses a threat of adversely affecting the water quality and quantity. Monitoring techniques provide means with which we can detect and characterize these impacts, and determine the biotic (i.e. biological) integrity of the specific water body. Biological integrity has been described as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organisation comparable to that of natural habitat of the region" (Karr and Dudley, 1981). The inefficiency of monitoring only the chemical-physical parameters (e.g., dissolved oxygen, pH, conductivity, turbidity, etc.) to determine the overall condition of the aquatic ecosystem is well documented (Worf, 1980; Hellawell, 1986; DWAF, 1997). Information concerning the biological components of the aquatic environment also needs to be acquired.

Biological assessment is defined as an evaluation of a water body using biological surveys and other direct measurements of the resident biota in surface waters (Barbour, 1997). It is thus possible to monitor a certain species, or indicator species, in order to determine the condition of a specific water body. Various taxonomic groups are used to assess water bodies, i.e. benthic macroinvertebrates and fish are often used to assess flowing waters, while plants are used in wetlands and algae and zooplankton in lakes and estuaries (DWAF, 1997).

Biological monitoring also provides unique information pertinent to the sustainable management of river basins (Chutter, 1995). It plays a vital role in generating data during the RDM methodology.

Inherent in the ERA framework is the stressor-response analyses that quantify the relationship between the stressor and the environmental value to be protected. Biological assessment and criteria fit well in this conceptual framework by providing a measurable representation of ecosystem integrity, quantifying environmental values to be protected, and responsiveness to the effects of non-chemical stressors (Davis and Simon, 1995).

In South Africa, the benthic macroinvertebrates have been more intensively studied than other components of the biota in relation to water quality (O'Keeffe, 1986). To address the need for information on the state of aquatic ecosystems in South Africa, DWAF has launched an initiative to develop a programme for monitoring the health of aquatic ecosystems (DWAF, 1997). This programme is known as the National Aquatic Ecosystem Biomonitoring Programme (NAEBP), and has been renamed to the River Health Programme (RHP). Invertebrates are one of the biological indicators that are considered appropriate for inclusion in the RHP, where invertebrate monitoring is performed by utilizing the SASS4 biological index. Taking the credibility and popularity of the SASS4 biomonitoring method into account, it was decided to investigate the role and application of this biological index in the ERA methodology. There is also aimed at determining how SASS4 data can be used to evaluate a facet of the RDM methodology that has been performed on the Luvuvhu River.

CHAPTER 2

ECOLOGICAL RISK ASSESSMENT

2.1 INTRODUCTION

Risk assessment can be defined as the process of assigning magnitudes and probabilities to the adverse effects of human activities or natural catastrophes (Suter, 1993). Ecological risk assessment is hence the process of estimating the probabilities of undesirable ecological events occurring and evaluating their consequences (Bartell, 1998). According to USEPA (1998), undesirable or adverse effects are viewed as those changes that alter important structural or functional characteristics or components of ecosystems.

It should be recognized, however, that as a component process used in a broader decision making context, ecological risk assessment includes qualitative aspects. For example, identifying and selecting ecological impacts to be assessed are often influenced by considerations of underlying social, political, and economic values relevant to the assessment (Bartell, 1998). Descriptions of the likelihood of adverse effects may also range from qualitative judgements to quantitative probabilities. Although risk assessments may include quantitative risk estimates, quantification of risks is not always possible (USEPA, 1998).

In 1992 the U.S. Environmental Protection Agency (EPA) published a report entitled *Framework for Ecological Risk Assessment*, which proposed principles and terminology for the ecological risk assessment process. The EPA further improved on this report and they published a new document entitled *Guidelines for Ecological Risk Assessment*, which was effective April 30, 1998. A typical schematic of the ERA process, as presented in these guidelines, is shown in Figure 2.1.

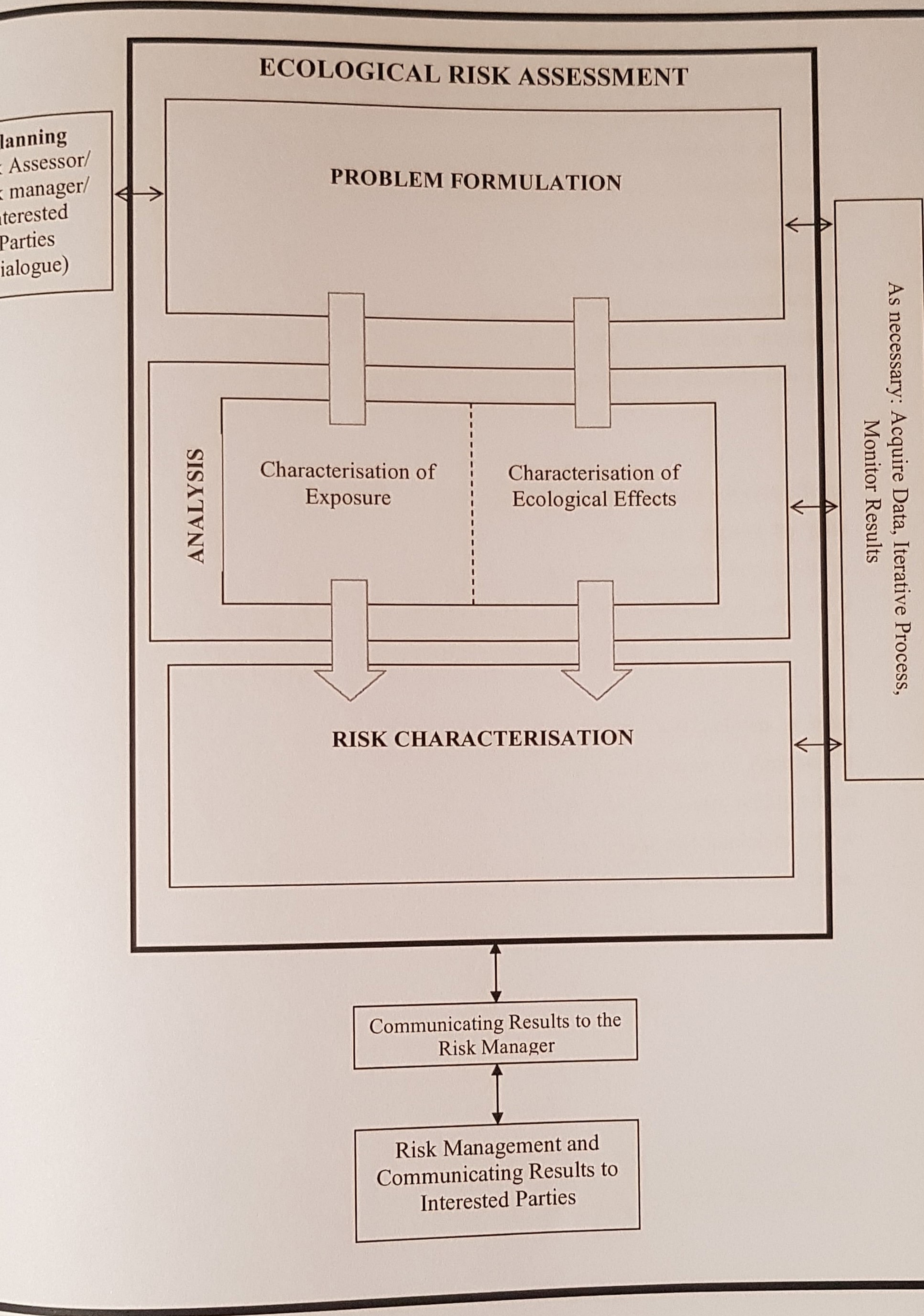

Figure 2.1: The Ecological Risk Assessment Framework

According to the USEPA (1998), Ecological Risk Assessment is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. The process is used to systematically evaluate and organize data, information, assumptions, and uncertainties in order to help understand and predict the relationships between stressors and ecological effects in a way that is useful for environmental decision making (USEPA, 1998). According to Roux *et al* (1997), a stressor is any physical, chemical or biological entity or process that can induce adverse effects on individuals, populations, communities or ecosystems. It should be noted that ecological risk assessment has been employed primarily to deal with chemicals (Suter, 1993). The *Guidelines* also primarily focus on human-induced impacts.

ERA can be performed in two ways: to predict the likelihood of future adverse effects (predictive ERA), or to evaluate the likelihood that effects are caused by past exposure to stressors (retrospective ERA). All predictive assessments begin with a proposed source (e.g., effluent), while the impetus for retrospective ERAs may be a source, observed effects, or evidence of exposure (Suter, 1993).

This chapter primarily focuses on the USEPA Guidelines, and delivers a brief overview of the ERA process as it is performed in the United States of America. It should be noted that the framework might vary according to the country within which an ERA is conducted. Taking this into consideration, a lot of the information to follow might be viewed as superfluous once ecological risk assessment is set in South Africa.

2.2 THE ECOLOGICAL RISK ASSESSMENT PROCESS

2.2.1 Planning the Risk Assessment

At the onset of an ERA, risk managers and risk assessors and in some cases interested parties, engage in a planning dialogue as a critical first step toward initiating problem formulation (USEPA, 1998).

Risk managers are responsible for ensuring that the necessary environmental management decisions can be supported by the results of the risk assessment (Murray and Claassen, 1999), and deciding what action will be taken (if required) to minimise the risk (NSW Environment Protection Authority, 1997).

In turn, risk assessors ensure that scientific information is effectively used to address ecological and management concerns (USEPA, 1998). They are the ones who actually undertake the assessment.

Interested parties (e.g. municipal, local, and national governments, industrial leaders, environmental groups) may also contribute to planning, where they communicate their concerns about factors that they consider as valuable, and that may be at risk.

During planning dialogues the characteristics of an ERA are determined. They include: (1) clearly established and articulated management goals, (2) characterization of decisions to be made within the context of the management goals, and (3) agreement on the scope, complexity, and focus of the risk assessment, including the expected output and the technical and financial support available to complete it (USEPA, 1998).

Management goals are statements about the desired condition of ecological values of concern (USEPA, 1998). Legislation serves as guidance for risk managers regarding what is to be protected (Barton and Sergeant, 1998). Management goals are achieved through the definition and implementation of management decisions. These decisions start of as management options, e.g. prevention of the introduction of a stressor, or the restoration of the affected ecological values.

Explicitly stated management options provide a framework for defining the scope, focus, and conduct of a risk assessment (USEPA, 1998). Agreement on the scope of an ERA includes gaining clarity on the constraints of data availability, scientific knowledge, financial resources, and spatial and temporal scales (Murray and Claassen, 1999).

Before the formal risk assessment process is undertaken, a summary report concerning the objectives that were agreed upon is produced.

2.2.2 Problem Formulation Phase

Problem formulation is a process for generating and evaluating preliminary hypotheses about why ecological effects have occurred, or may occur, from human activities (USEPA, 1998). This critical step establishes the direction and scope of the ecological risk assessment (Menzie and Freshman, 1997).

Problem formulation generates three products: (1) assessment endpoints, (2) conceptual models, and (3) an analysis plan. The first step toward developing these products is to integrate available information.

The problem formulation phase is started with the integration and evaluation of available information on stressor sources and characteristics, exposure opportunities, characteristics of the ecosystem(s) potentially at risk, and ecological effects. Knowledge gained during this integration is used to identify missing information and potential endpoints. It also contributes to the early conceptualization of the potential impact.

2.2.2.1 Selecting Assessment Endpoints

Assessment endpoints are explicit statements of the environmental values to be protected (Cook *et al*, 1999). Assessment endpoints structure the assessment to address management concerns and form the basis for the development of the conceptual model. According to Barton and Sergeant (1998), an assessment endpoint constitutes an entity or valued resource that is to be protected, and an attribute or aspect of that entity

Ecological values to serve as assessment endpoints are chosen according to the following criteria: (1) ecological relevance, (2) susceptibility to known or potential stressors, and (3) relevance to management goals.

Ecologically relevant endpoints reflect important characteristics of the system and are functionally related to other endpoints (USEPA, 1998). Ecologically relevant endpoints may be identified at any level of organization (e.g., individual, population, community, ecosystem, and landscape). In specific cases professional judgment based on site-specific information, preliminary surveys, or other available information is applied in determining ecological relevance.

Ecological resources are considered susceptible when they are sensitive to a stressor to which they are, or may be exposed to (USEPA, 1998). Susceptibility is often identified early in problem formulation, but sometimes selection requires professional judgment.

Sensitivity refers to how readily an ecological entity is affected by a particular stressor, and is directly related to the stressor's mode of action (USEPA, 1998). Individual and community life-history characteristics, the life stage of an organism during exposure, and the presence of other stressors or natural disturbances also influence sensitivity. Mortality, adverse reproductive effects, or behavioral abnormalities are all considered as measures of sensitivity.

In order to take into account exposure, during the estimation of susceptibility, it is important that the assessor considers the proximity of an ecological value to stressors of concern, the timing of exposure (both in terms of frequency and duration), and the intensity of exposure occurring during sensitive periods (USEPA, 1998).

It is important that risk managers and the public also recognize the ecological entities that are chosen as assessment endpoints as valuable. This will ensure that the risk assessment contributes to management decisions.

Once ecological values are selected as potential assessment endpoints, they need to be operationally defined. Two elements are required to define an assessment endpoint. The first is the identification of the specific valued ecological entity. This can be a species (e.g., eelgrass), a functional group of species (e.g., piscivores), a community (e.g., benthic invertebrates), an ecosystem (e.g., lake), a specific valued habitat (e.g., wetland), or other entity of concern (USEPA, 1998). The second is the characteristic about the entity of concern that is important to protect and potentially at risk. An example is the abundance and diversity (characteristic) of benthic invertebrate (ecological entity) species.

2.2.2.2 Conceptual Models

A conceptual model in problem formulation is a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed (USEPA, 1998).

Conceptual models consist of two principal components:

1. A set of risk hypotheses that describe predicted relationships among stressor, exposure, and assessment endpoint response, along with the rationale for their selection
2. A diagram that illustrates the relationship presented in the risk hypotheses.

Risk hypotheses are specific assumptions about potential risk to assessment endpoints and may be based on theory and logic, empirical data, mathematical models, or probability models (USEPA, 1998). They are formulated using a combination of professional judgment and available information on the ecosystem at risk, potential sources of stressors, stressor characteristics, and observed or predicted ecological effects on selected or potential assessment endpoints.

Diagrams in a conceptual model are a visual representation of risk hypotheses. Typical conceptual model diagrams are flow diagrams containing boxes and arrows to illustrate relationships. A diagram's usefulness however, is linked to the detailed written descriptions and justifications for the relationships shown. When developing conceptual model diagrams, factors to consider include the number of relationships depicted, the comprehensiveness of the information, the certainty surrounding a linkage, and the potential for measurement (USEPA, 1998).

Problem formulation should end off with a summary of the description of the nature of the uncertainties that are encountered throughout this phase. Uncertainty is a lack of confidence in the prediction of a risk assessment that may result from natural variability in natural processes, imperfect or incomplete knowledge, or errors in conducting an assessment (Society of Environmental Toxicology and Chemistry, 1997)).

2.2.2.3 Analysis Plan

The analysis plan is the final stage of problem formulation. During analysis planning, risk hypotheses are evaluated to determine how they will be assessed. The plan includes methods for conducting the analysis phase of the risk assessment (USEPA, 1998).

The analysis plan includes pathways and relationships identified during problem formulation that will be pursued during the analysis phase. Those hypotheses considered more likely to contribute to risk are targeted.

There are various measures that should be selected at this point (Murray and Claassen, 1999). One group consists of measures of effect that evaluate the response of the assessment endpoint when exposed to a stressor. Another constitutes measures of exposure, which establish mechanisms by which exposure occurs. A third group comprises measures of ecosystem and receptor characteristics, which affect the assessment endpoints.

Finally the analysis plan is reviewed by the risk manager and the risk assessor to ensure that the plan will influence decision making.

The plan should clearly identify the data that needs to be measured and the information that needs to be collated (Murray and Claassen, 1999).

2.2.3 Analysis Phase

The analysis phase examines the two primary components of risk, namely exposure and effects, and their relationships between each other and ecosystem characteristics.

During this process, the risk assessor is responsible for selecting data, analyzing exposure, analyzing effects, and summarizing the conclusions about exposure and effects.

2.2.3.1 Evaluating Data and Models for Analysis

Here the risk assessor must first critically evaluate existing studies. The strengths and limitations of data from various sources must be established (Murray and Claassen, 1999). These sources include laboratory and field studies, indices, experience from similar situations, structure-activity relationships and models. Studies should be evaluated for their utility in risk assessment, their purpose and scope, and their design and implementation. It should be noted however that there is no universal method for quantifying ecological risks that will produce precise, general, and realistic results (Suter, 1993).

Uncertainty evaluation is the following step, where the objective is to describe and, preferably, quantify the known and unknown about exposure and effects. Sources of uncertainty include unclear communication, descriptive errors, data gaps, uncertainty about a quantity's true value, model structure uncertainty, and uncertainty about a model's form (USEPA, 1998).

2.2.3.2 Characterization of Exposure

In short, the exposure analysis asks about the potential sources of exposure for environmentally exposed populations, the chemical or physical form of the stressor, its transformations in time and space, and its bioavailability (Patton, 1998).

All of this information is then integrated into the exposure profile - a summary of the results of the exposure analysis.

2.2.3.3 Characterization of Ecological Effects

The first step in ecological response analysis is to examine the stressor-response relationship. There should be a correlation between the effects and the assessment endpoints and the conceptual model. A response variable (e.g. mortality) is analyzed using quantitative techniques, although qualitative evaluations are also possible (e.g. high, medium, and low) (USEPA, 1998).

The causality (i.e. the relationship between cause and effect) is then established. Evidence is needed to link cause and effect. General criteria can be used to affirm (e.g. strength of association) or reject (e.g. inconsistency in association) causality

Following this, the measures of effect are linked to the assessment endpoints. If it is difficult to measure the assessment endpoints directly risk assessors may use extrapolation methods to link measures of effect to assessment endpoints (USEPA, 1998). It is important for these linking methods to be consistent with ecological principles, and to use enough appropriate data. Linking may be based on professional judgment when there is a lack in data, or it may be based on empirical or process models.

Finally a stressor-response profile (a summary of the above) is compiled.

2.2.4 Risk Characterization

Risk characterization is the product of the risk assessment (Patton, 1998). The main objective of this final phase is to integrate the exposure and effects information into an understanding of the ecological risks, followed by a risk description. The associated uncertainties should also be evaluated. The conclusions are then presented to the risk managers.

The USEPA (1998) identifies the following techniques to estimate risk:

1. The first technique is to use field observational studies, where the risk is determined directly from the results.
2. Risks can also be ranked using categories (e.g. low, medium, and high) which would serve as a qualitative evaluation.
3. Another method is to use a ratio (or quotient) to compare exposure and effects estimates, where the ratio is expressed as an exposure concentration divided by an effects concentration.
4. A stressor-response curve may be compared with an exposure distribution. With this technique risk estimation can examine risks associated with many different levels of exposure.
5. The next method is to incorporate variability in exposure and/or effects. With exposure, this would allow one to estimate risks to moderately or highly exposed population members, whereas with effects, it can be used to estimate risks to average or sensitive members of a population.
6. Finally, risk estimates may be based on the application of process models (i.e. mathematical expressions that represent our understanding of the mechanistic operation of a system). These models supply point estimates, distributions, or correlations.

During the risk description two evaluations are performed. Firstly, lines of evidence have to be developed to increase the confidence of the risk estimate. Data quality, the degree and type of uncertainty, and the relationship of the results to the risk assessment hypotheses have to be considered (USEPA, 1998). The next step is to interpret the significance of the adverse ecological effects on the assessment endpoints. Criteria for determining ecological adversity include the nature and intensity of effects, the spatial and temporal scale, and the recovery potential (USEPA, 1998).

The risk assessment is concluded with a risk assessment report. This report may include: risk assessor/risk manager planning results, revision of the conceptual model and assessment endpoints, major data sources used, revision of stressor-response and exposure profiles, risks to assessment endpoints, and a revision and a summary of major areas of uncertainty.

2.2.5 Influencing Risk Management Decisions

Risk management is the process of selecting and implementing a strategy for control of a risk, followed by monitoring and evaluation of the effectiveness of that strategy (Kwiatkowski, 1998). On completion of the risk assessment report, the risk assessors discuss the results with the risk managers. In addition to these results, the risk managers consider economic, legal, political, and social issues to influence decision-making. The communication of the risk information to the public and interested parties should include a description of the risk source and its potential effects, and it is also important to answer particular questions of specific individuals, and results should thus be presented in a clear and understandable format/way (USEPA, 1998).

The risk assessment - the process for determining the extent of the risk - should be separate and distinct from the risk management - the mechanism for evaluating the feasibility and costs of the controls (Cotruvo, 1987). The reason for this being that the assessment is characterized by many uncertainties, and management decisions must be made in the light of those uncertainties as well as taking into account economic and technological realities and social demands.

If additional follow-on activities are required, they should be identified during the risk management process. If the risk assessment fails to answer important questions the risk manager may choose to conduct an iteration (a re-evaluation of information) of the risk assessment.

Final management decisions should be monitored to determine whether mitigation efforts, source reduction, or ecological recovery is achieved. This will serve as an evaluation of the effectiveness of the ecological risk assessment.

In Chapter 6, the ERA methodology is further examined by determining how SASS4 contributes to the execution of this process.

CHAPTER 3

BIOMONITORING

3.1 INTRODUCTION

What is the importance of monitoring our water resources? Humans have realized that their well being is irrevocably linked to the health of aquatic ecosystems. We utilize this resource in numerous ways in our daily lives. Unfortunately, we also have a diversity of impacts on water systems, which affect the water quality and quantity. Monitoring techniques provide means with which we can detect and characterize these impacts.

In the past, most pollution monitoring programs trusted chemical-physical parameters (e.g., dissolved oxygen, pH, conductivity, turbidity, etc.) to evaluate the condition of a water body. It was however realized that these methods were insufficient to assess the health of an aquatic system (Worf, 1980; Hellawell, 1986; DWAF, 1997). Chemical monitoring covers only a fraction of the possible toxins that may be present in water and the chemical analysis process takes a relatively long time in comparison with the reaction time of organisms. Further, chemical monitoring doesn't take into account many man-induced disturbances (e.g., flow alterations), nor short-term pollution-induced stresses. Conveniently, aquatic organisms serve as integrators of their total environment, and their response to complex sets of environmental conditions are used as monitors of water quality (Worf, 1980).

According to Larsen (1997), biological assessment of aquatic ecosystems are intended to produce information about the condition of water resources by examining the density and relative abundance of resident organisms, the condition of their immediate habitat (e.g., physical habitat structure, water quality, hydrology) and the condition of their watershed. Biological assessment can thus be defined as an evaluation of the condition of a water body using biological surveys and other direct measurements of the resident biota in surface waters (Barbour, 1997).

Development and implementation of biological monitoring programs are important to set planning and management priorities for water bodies in South Africa most in need of control (Heath, 1993). Figure 3.1 illustrates the role of biological monitoring in water quality management.

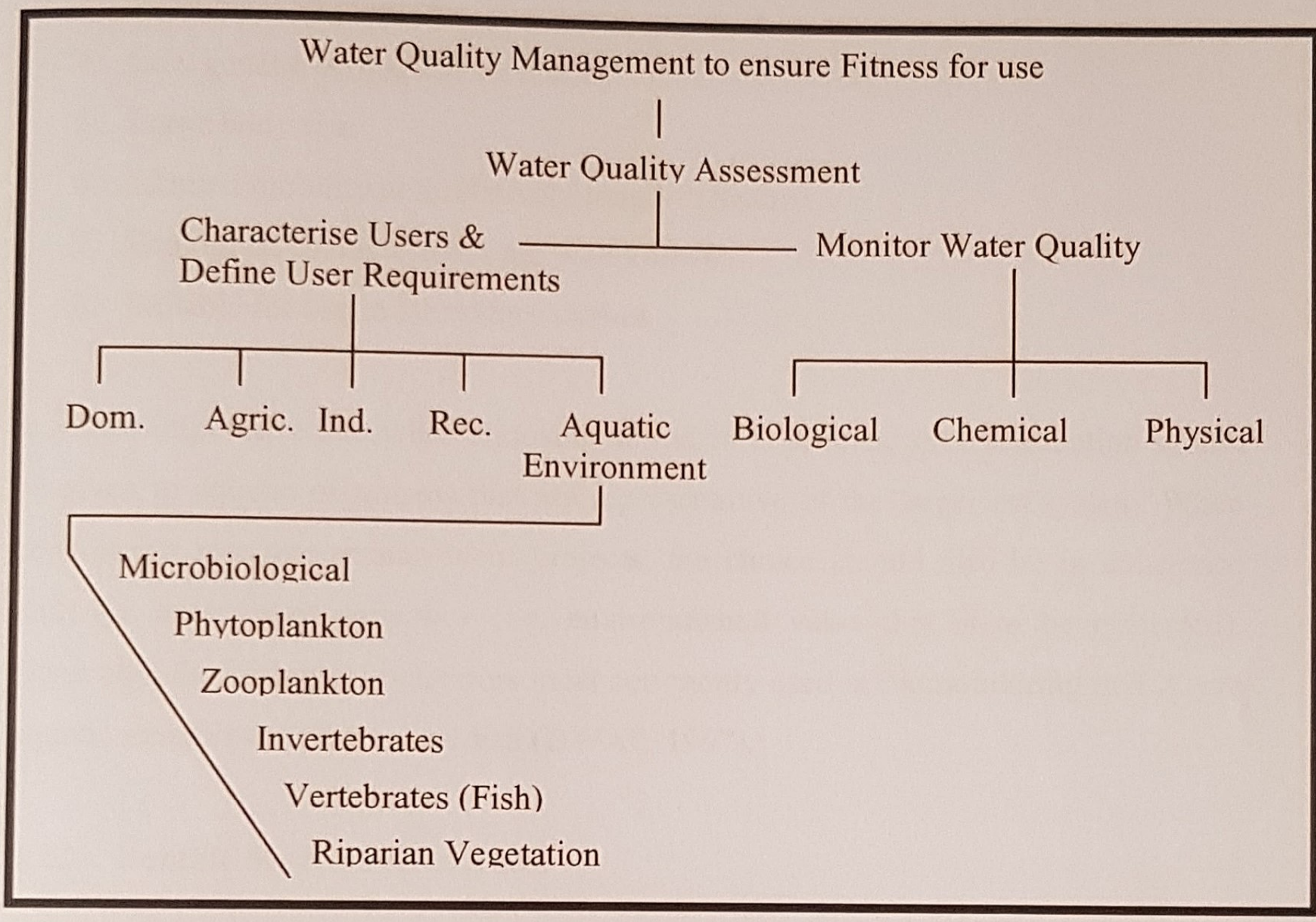

Figure 3.1: The role of biological monitoring in water quality management (Heath, 1993).

3.2 BIOMONITORING

3.2.1 Indicator Species

Any pollution arriving into the environment influences the ecosystem through individuals of the microbial, plant and animal population (Salanki, 1986). It is thus possible to monitor a certain species, or indicator species, in order to determine the condition of a specific water body. Indicator species are those organisms that are generally known to respond to environmental contaminants in particular ways, based on scientifically supportable observations (Stahl, 1997).

According to Rosenberg and Resh (1993), the "ideal" indicator should have the following characteristics:

1. Taxonomic soundness and easy recognition by the nonspecialist
2. Cosmopolitan distribution
3. Numerical abundance
4. Low genetic ecological variability
5. Large body size
6. Limited mobility and relatively long life history
7. Ecological characteristics are well known
8. Suitable for use in laboratory studies

It is thus imperative that when choosing an indicator species, special attention should be given to aquatic organisms that are representative of the larger ecosystem. When considering resource management projects, the choice should also be in coherence with the assessment endpoints (i.e. environmental value that is to be protected). Generally, the biological indicators most commonly used in biomonitoring in S.A. are aquatic macroinvertebrates and fish (DWAF, 1997).

3.2.2 Benthic Macroinvertebrates

The choice of benthic invertebrates for evaluation of the quality of surface waters has long been recognized as one of the most valuable tools for monitoring aquatic ecosystems (Worf, 1980). In South Africa, the benthic macroinvertebrates have been more intensively studied than other components of the biota in relation to water quality (O'Keeffe, 1986).

Benthic macroinvertebrates (benthic = bottom, macro = large, invertebrate = animal without a backbone) refers to organisms that inhabit the bottom substrates (sediment, debris, logs, macrophytes, filamentous algae, etc.) of freshwater habitats, for at least part of their life cycle (Rosenberg and Resh, 1993).

The two most common types of biomonitoring using benthic macroinvertebrates include surveillance and to ensure compliance (Rosenberg and Resh, 1993). The first approach includes surveys done before and after an impact has occurred, or to survey if water resource management techniques are working. The second use is ensure that immediate statutory requirements are met, or to control long-term water quality.

Benthic macroinvertebrates exhibit certain responses when confronted with adverse surroundings. According to Rosenberg and Resh (1993), these responses originate at the biochemical and physiological levels of an individual organism, and two groups of invertebrates have even showed morphological deformities: the Insecta and the Oligochaeta. Jeffrey and Madden (1991) noted the following behavioral responses to pollutants, in certain freshwater macroinvertebrates:

- a decrease in the case building ability of the caddis fly (*Trichoptera*) *Agapetus fuscipes,*
- a depression of feeding rate in the freshwater amphipod *Gammarus pulex,* and
- a change in the reproduction behavior of the midge *Chironomus riparius.*

Salanki (1986) found similar anomalies and responses. Life-history indicators of environmental stress in freshwater macroinvertebrates include survival, growth, and reproduction. According to Mokgalong (1981), there is also a possible correlation between chemo-physical parameters and invertebrate drift occurrences, where drift refers to the downstream transportation of stream-dwelling organisms in the water column (Allan, 1995).

Benthic macroinvertebrates have several characteristics that make them advantageous for use in bioassessments (Voshell *et al*, 1997):

1. They occur in almost all types of freshwater habitats;
2. There are many different taxa of benthic macroinvertebrates, and among these taxa there is a wide range of sensitivity to all types of pollution and environmental stress;
3. Benthic macroinvertebrates have mostly sedentary habits so they are likely to be exposed to pollution or environmental stress;
4. The duration of their life history is sufficiently long that they will likely be exposed to pollution and environmental stress, and the assemblage will not recover so quickly that the impairment will go undetected;

5. Sampling the benthic macroinvertebrate assemblage is relatively simple and does not require complicated devices or great effort; and
6. Taxonomic identification is almost always easy to the family level and usually relatively easy to the genus level.

Of particular importance, in relation to biomonitoring, are the differences in sensitivity and tolerances to pollution between the different invertebrate groups.

Unfortunately, there are also difficulties when using benthic macroinvertebrates in biomonitoring, which according to Rosenberg and Resh (1993), are as follows. Not all impacts effect benthic macroinvertebrates. Secondly, water quality is not the only factor that influences their abundance and distribution. Natural conditions (e.g. substrate type) also play a role. Thirdly, sampling problems are created by seasonal variations in abundance and distribution. Finally, macroinvertebrates may be carried into areas where they don't normally occur. These difficulties may be overcome through knowledge of the life history, habitat preferences, and drifting behavior of the species involved.

Biological surveillance of communities - with special emphasis on characterizing taxonomic richness and composition - is perhaps the most sensitive tool now available for quickly and accurately detecting alterations in aquatic ecosystems (Rosenberg and Resh, 1993). It is thus more advantageous to look at the structure of an invertebrate community, when assessing water quality based on macroinvertebrate indicators, than to examine individual taxa. The structure of a biotic community includes diversity, richness, and interspecific associations.

It is also possible to assess the biotic community function, where one would look at the productivity processes, decomposition, and energy and nutrient fluxes. According to DWAF (1999), the major roles of invertebrates in river functioning can be summarised as:

- Retention and breakdown of organic material;
- Recycling of minerals and nutrients; and
- Contributions to energy processing in the river at different trophic levels.

Biomonitoring of benthic macroinvertebrates can also provide insight into the nature of the stream disturbance through an examination of the predominant functional feeding groups of macroinvertebrates present. For example, an increase in the number of collectors may indicate organic enrichment. According to Townsend (1980), benthic macroinvertebrates are divided into four functional feeding groups:

(a) Grazers - herbivores feeding on attached algae.

(b) Shredders - organisms feeding on large particles of plant material.

(c) Collectors - organisms feeding on fine particles either on the stream bed or filtered from the water.

(d) Predators - organisms that feed directly on other aquatic animals such as fish and invertebrates.

Furthermore, as a river progresses each reach is dominated by invertebrates that have feeding habits that are characteristic of the sizes of the particulate matter that dominate in that stretch of the river. The upper parts of the river will have coarse particulate organic matter, which will then be broken down to finer material that then enters the middle reaches. In the lower reaches the material will be predominately very fine, and will settle out of the water column as the current slows. These changes are in accordance with the river continuum concept, where biological adjustment are evident in (a) the changing balance of production and decomposition (the ratio of photosynthesis:respiration) and (b) in changes in community composition, expressed as a downstream succession of "functional feeding groups": the shredders, grazers, collectors (O'Keeffe, 1986). The distribution of invertebrates down the length of a river is also influenced by abiotic factors such as current, substratum, oxygen and temperature, and concentrations of dissolved chemicals.

It has been mentioned previously that benthic macroinvertebrates are suitable indicators of impacts. For example, when toxicants are added to water a chemical analysis won't reflect the true impact as the chemicals will be washed downstream, while there will be a drastic change in the invertebrate community for quite a time after the chemicals have vanished. It has also been found that even at a distance of 1.5 km downstream from a trout farm, the effluent still influences taxa richness (Loch *et al*, 1996). How long does it however take for an invertebrate community to recuperate and return to normal, and through what processes? An example of the return to normality was determined by Muirhead-Thomson (1987), where invertebrates showed a remarkably rapid recovery after a community was treated with the insecticide methoxychlor, e.g. *Chironomidae* took 1 - 2 weeks, *Trichoptera* 1 - 3 weeks, and *Plecoptera* 4- -5 weeks. Recovery also occurs when river currents transport upstream invertebrates to the disturbed areas. Through this process, known as drifting, there is thus a redistribution of invertebrates, but permanent residency can only be established once suitable conditions exist.

3.2.3 Substrate Influence

The great majority of stream-dwelling macroinvertebrates live in close association with the substrate, and many taxa show some degree of substrate specialisation. The main factor that restricts occupation is the substrate particle size, which determines the size of the interstitial spaces which, in turn, affects the type of organisms comprising the bottom-dwelling community (Dallas, 1995). Table 3.1 indicates how inorganic substrates are classified.

Very small organic particles (less than 1 mm) usually serve as food rather than as substrate, whereas larger organic material, from plant stems to submerged logs, generally functions as substrate rather than food.

3.2.4 South African Scoring System (SASS)

To address the need for information on the state of aquatic ecosystems in South Africa, the DWAF has launched an initiative to develop a programme for monitoring the health of aquatic ecosystems (DWAF, 1997). This programme is known as the National Aquatic Ecosystem Biomonitoring Programme (NAEBP). The overall objective of the riverine programme, renamed the River Health Programme (RHP), is to develop the procedures and infrastructure for implementation and ongoing maintenance of biomonitoring on a national scale (DWAF, 2000). Invertebrates are one of the biological indicators that are considered appropriate for inclusion in the RHP.

Table 3.1: The classification of mineral substrates by particle size, according to the Wentworth Scale (Allan, 1995).

Size Category	Particle Diameter (range in mm)
Boulder	>256
Cobble	
Large	128-256
Small	64-128
Pebble	
Large	32-64
Small	16-32
Gravel	
Coarse	8-16
Medium	4-8
Fine	2-4
Sand	
Very coarse	1-2
Coarse	0.5-1
Medium	0.25-0.5
Fine	0.125-0.25
Very fine	0.063-0.125
Silt	<0.063

A biological/biotic index has been developed for South African aquatic conditions, based on the composition of aquatic invertebrate communities. This method is a modification from the BMWP (Biological Monitoring Working Party) scoring system that is used in England, and it has been named SASS (South African Scoring System) (Chutter, 1998).

Score systems such as the BMWP assign scores to biotic groups based on generally accepted organism sensitivities to pollution and habitat disturbances (i.e., stoneflies, caddisflies, and mayflies are given high scores based on their presence and abundance). Various indices have been used to determine the change in species composition, e.g. Saprobic Index, Trent Biological Index, Chandler Score (Loch *et al*, 1996).

SASS is thus a scoring system based on benthic macroinvertebrates, whereby each taxon is allocated a sensitivity/tolerance score according to their susceptibility to changes in water quality conditions (Dallas, 1997).

The SASS has undergone changes since its introduction in this country. The initial SASS scoring system was documented by Moore and McMillan (1992) and was known as SASS2. It finally evolved into the method known today as SASS4 (South African Scoring System version 4).

An example of the SASS4 score sheet is given in Table 3.2. Information required includes the sampling locality, river name, date, and biotopes sampled. Habitats or the biotopes to be sample, include stones out of current (SOOC), stones in current (SIC), sand, gravel, mud, marginal vegetation, and aquatic vegetation. Benthic macroinvertebrates have different habitat preferences, and sampling all the available biotopes ensures that a true representation of the community, at a specific locality is obtained.

Biotopes can further be grouped into specific biotopes to provide details of the types of biotopes within each SASS biotope. According to DWAF (2000), the specific biotopes for each SASS biotope is as follows:

- SIC - cobble riffle, run, bedrock rapid, chute, cascade, and waterfall.
- SOOC - backwater, slackwater, and pool.
- Marginal vegetation - grasses, reeds, shrubs, sedges, etc. (i.e., vegetation adjacent to the river bank).
- Aquatic vegetation - sedges, trailing grasses, etc. (i.e., vegetation that is in the channel, submerged or partially submerged.

- Gravel, sand and silt/mud/clay.

Different families are also allocated different scores according to their sensitivity to deteriorating water quality. High numbers are thus associated with greater sensitivity and low scores with greater tolerances. From the completed sample analysis sheet the scores are summed to give a sample score, and the number of families are determined. Finally the Average Score Per Taxon (ASPT) is calculated by dividing the sample score with the number of families found.

3.2.5 Habitat Assessment

In order for a bioassessment to obtain results that adequately reflect the true condition of a particular stream or river, a habitat assessment needs to be performed. According to Kleynhans (1996), the assessment of the habitat integrity (i.e. the maintenance of a balanced, integrated composition of physico-chemical and habitat characteristics on a temporal and spatial scale that are comparable to the characteristics of natural habitats of the region) of a river can be seen as a precursor of the assessment of biotic integrity.

According to Heath (1993), the supporting role of habitat assessment in biosurveys include the following:

- assists in the selection of appropriate sampling sites;
- provides basic information for interpreting biosurvey results; and
- is used to identify obvious constraints on the attainable potential of a specific site.

Wright *et al* (1998) also realized the importance of including a River Habitat Survey (RHS) when using invertebrates in the classification of rivers and the development of the River Invertebrates Prediction And Classification System (RIVPACS).

Benthic macroinvertebrates not only differ in their sensitivity/tolerances to different pollutants, but also in their habitat preferences. Given that certain taxa are commonly associated with a particular biotope, it seems likely that the number and types of biotopes available for habitation by aquatic biota, and which are thus sampled by

SASS4

River................. ....Date.................Time.................

Sampling point..

Temp..................pH..................EC(mS/m)......................

DO(mg/l)................% sat.................Turb....................

Biotopes sampled

SIC.............(Type./time...)

Marg veg............Dom.sp..

Aq veg............Species...

SOOC............Sand........... Mud........... Gravel............

Other...HABS1.....................

Procedure Protocols

If SIC all kickable, sample for 2 min., otherwise for maximum of 5 min.
Gravel 1/2 min.
Marg/Aq veg, back & forward sweep 2 m.
SOOC kick +- 1m
Sand/mud stir with feet & sweep net over disturbed area for 1/2 min.
Any other biotopes -1/2 min.
Complete top of form.
Tip net contents into tray. Remove leaves, twigs & trash.
Check taxa present FOR THE LESSER of 15 minutes or 5 minutes since the last taxon was found.
Estimate abundance on scale: A 1-10; B 11-100 C 100 - 1000; D >1000
BEFORE LEAVING THE SAMPLING POINT CHECK

TAXON	SCORE	ABUN
Porifera	5	
Coelenterata		
Hydra sp.	1	
Turbellaria		
Planarians	5	
Annelida		
Oligochaeta	1	
Hirudinea	3	
Crustacea		
Amphipoda	15	
Crabs*	3	
Shrimps	8	
Hydracarina		
Hydrachnellae	8	
Plecoptera		
Notonemouridae	12	
Perlidae	12	
Ephemeroptera		
Polymitarcyidae	10	
Ephemeridae	15	
Baetidae 1 sp	4	
2 spp	6	
> 2 spp	12	
Oligoneuridae	15	
Heptageniidae	10	
Leptophlebiidae	13	
Ephemerellidae	15	
Tricorythidae	9	
Prosopistomatidae	15	
Caenidae	6	
Odonata		
Chlorolestidae	8	
Lestidae	8	
Protoneuridae	8	
Platycnemidae	10	
Coenagriidae	4	
Calopterygidae	10	
Chlorocyphidae	10	
Zygoptera juvs	6	
Gomphidae	6	

Hemiptera	SCORE	ABUN
Notonectidae*	3	
Pleidae*	4	
Naucoridae*	7	
Nepidae*	3	
Belostomatidae*	3	
Corixidae*	3	
Gerridae*	5	
Veliidae*	5	
Megaloptera		
Corydalidae	8	
Trichoptera		
Hydropsychidae 1 spp	4	
2 spp	6	
> 2 spp	12	
Philopotamidae	10	
Polycentropodidae	12	
Psychomyiidae	8	
Ecnomidae	8	
Hydroptilidae	6	
Other movable case larvae:		
case types score fam		
1 8 1	8	
2 15 1	15	
3 20 1	20	
4 30 2	30	
5 40 2	40	
>5 50 3	50	
Lepidoptera		
Nymphulidae	15	
Coleoptera		
Dytiscidae (adults*)	5	
Elmidae/Dryopidae	8	
Gyrinidae (adults*)	5	
Haliplidae (adults*)	5	
Helodidae	12	
Hydraenidae (adults*)	8	
Hydrophilidae (adults*)	5	
Limnichidae	8	
Psephenidae	10	

Diptera	SCORE	ABUN
Blepharoceridae	15	
Tipulidae	5	
Psychodidae	1	
Culicidae*	1	
Dixidae*	13	
Simuliidae	5	
Chironomidae	2	
Ceratopogonidae	5	
Tabanidae	5	
Syrphidae*	1	
Athericidae	13	
Empididae	6	
Ephydridae	3	
Muscidae	1	
Gastropoda		
Lymnaeidae*	3	
Melaniidae*	3	
Planorbidae*	3	
Physidae*	3	
Ancylidae	6	
Hydrobidae*	3	
Pelecypoda		
Sphaeridae	3	
Unionidae	6	
Sample score		
No. of families		
Score/taxon (ASPT)		
Air breathers fam.		
Air breathers score		

Other families present

SASS, may affect scores (Dallas, 1997). It should be noted however that where there is severe pollution, habitat availability is not a factor in the SASS4 score achieved, considering that most of the really tolerant taxa are found in most habitats, e.g. chironomids.

The SASS4 user manual (Thirion *et al*, 1995) identifies three habitat indices that may be used in conjunction with SASS4: the HABS1 habitat assessment, the Habitat Assessment Matrix (HAM), and the Habitat Quality Index (HQI).

In short, explanations of these habitat indices are as follows:

- **HABS1**

The habitat is assessed based on the nature of the biotopes available at the sampling locality.

- **HAM**

It focuses on the impact of physical habitat degradation on a SASS score.

- **HQI**

This index is similar to the HAM.

McMillan (1998) has developed the Invertebrate Habitat Assessment System (IHAS) to incorporate habitat variability as an influencing factor in deriving SASS scores. An example of the IHAS version 2 score sheet is given in Table 3.3. This form is divided into two sections: the sampling habitat, and the stream characteristics. In the first section, each type of sampling habitat is allocated an 'ideal' value, according to availability of biotopes and the condition of the stream. This 'ideal' value is shown in bold on the score sheet. Adjustment numbers are calculated by subtracting the score of each subsection from the maximum score (20, 15, and 20 for each subsection respectively). A total adjustment score is then obtained and added to the SASS4 score. The second section notes the physical characteristics of the stream. The original SASS score still stands as the official figure for a specific locality; the modified figure should be used only to compare different localities, or possibly project values to 'ideal' sampling conditions (McMillan, 1998).

3.2.6 Water Quality

Adverse changes in biological communities may be attributed either to deterioration in water quality or to habitat degradation, or to both (DWAF, 1997). Water quality variables potentially affecting riverine ecosystems may be physical (turbidity, suspensoids, temperature) or chemical (non-toxic: pH, TDS, conductivity, individual ions, nutrients, organic enrichment and dissolved oxygen; and toxic: biocides and trace metals) (Dallas and Day, 1993).

Water quality can be influenced by catchment characteristics. According to Heath (1999), the following biophysical features of a catchment influence water quality:

- Topography;
- Climate;
- Geology;
- Soils; and
- Land use.

The presence or absence of certain taxa can be an indication of what type of pollution is present, e.g. a community dominated by midge larvae of the genus *Chironomus* can reflect an area with low-dissolved oxygen concentrations and high organic enrichment. The use of linking certain invertebrate community characteristics to accompanying aquatic conditions will be discussed further in Chapter 8.

able 3.3: IHAS version 2 score sheet (McMillan, 1998).

INVERTEBRATE HABITAT ASSESSMENT SYSTEM (IHAS)

version 2.2 River Name:

Site Name:

Date:

SAMPLING HABITAT

ones in Current (SIC)	0	1	2	3	4	5
al length of white water rapids (ie: bubbling water) (in metres)	none	0-1	>1-2	>2-3	>3-5	>5
al length of submerged stones in current (run) (in metres)	none	0-2	>2-5	>5-10	>10	
nber of separate SIC *area's* kicked (not individual stones)	0	1	2-3	4-5	6+	
rage stone size's kicked (cm's)(<2 *or* >20 is '<2>20')(gravel is <2; bedrock is >20).	none	<2>20	2-10	11-20	2-20	
ount of stone surface clear (of algae, sediment etc.) (in percent %) *	n/a	0-25	26-50	51-75	>75	
TOCOL: time spent actually kicking SIC's (in minutes)(gravel/bedrock = 0 min)	0	<1	>1-2	2	>2-3	>3

(* *NOTE:* up to 25% of stone is usually embedded in the stream bottom)

max. 20

SIC Score:

getation	0	1	2	3	4	5
gth of fringing vegetation sampled (river banks) (PROTOCOL - in metres)	none	0-½	>½-1	>1-2	2	>2
unt of aquatic vegetation/algae sampled (underwater) (in square metres)	none	0-½	>½-1	>1		
ging vegetation sampled in: ('still'=pool/still water only; 'run'=run only)	none		run	still		mix
e of veg. (percent leafy veg. as opposed to stems/shoots) (aq. veg. only=49%)	none		1-25	26-50	51-75	>75

max. 15

Vegetation Score:

er Habitat / General	0	1	2	3	4	5
es Out Of Current (SOOC) sampled: (PROTOCOL - in square metres)	none	0-½	>½-1	1	>1	
l sampled: (PROTOCOL – in minutes) ('under' = present, but only under stones).	none	under	0-½	>½-1	1	>1
sampled: (PROTOCOL – in minutes) ('under' = present, but only under stones).	none	under	0-½	½	>½	
el sampled: (PROTOCOL – in minutes) (*if all gravel, SIC stone size = '<2'*)**	none	0-½	½	>½**		
ock sampled: ('all'=no SIC, sand, or gravel; *then SIC stone size ='>20'*)**	none	some			all**	
presence: ('1-2m²'=algal bed; 'rocks'=on rocks; 'isol.'=isolated clumps) ***	>2m²	rocks	1-2m²	<1m²	isol.	non
identification: (PROTOCOL – using time: 'corr' = correct time)		under		corr		ove

(** *NOTE:* you must still fill in the SIC section)

max. 20

Other Habitat Score:

max. 55

Habitat Total:

STREAM CONDITION

ical	0	1	2	3	4	5
make up: ('pool'=pool/still/dam only; 'run' only; 'rapid' only; '2mix'=2 types etc.)	pool		run	rapid	2 mix	3 m
ge width of stream: (metres)		>10	>5-10	<1	1-2	>2-
ge depth of stream: (metres)	>1	1	>½-1	½	<½-¼	<¼
ximate velocity of stream: ('slow'=<½m/s; 'fast'=>1m/s) (use twig etc. to test).	still	slow	fast	med.		mi
r colour: ('disc.'=discoloured with visible colour but still transparent)	silty	opaque		disc.		cle
t disturbances due to: ('constr.'=construction; 'fl/dr'=flood or drought) ***	fl/dr	fire	constr.	other		nor
/ riparian vegetation is: ('grass'=includes reeds; 'shrubs'=includes trees)	none		grass	shrubs	mix	
unding impacts: ('erosn'=erosion/shear bank; 'farm'=farmland/settlement)***.	erosn.	farm	trees	other		ope
bank cover (rocks and vegetation): (in percent %)	0-50	51-75	75-95	>95		
bank cover (rocks and vegetation): (in percent %)	0-50	51-75	75-95	>95		

(*** *NOTE:* if more than one option, choos ethe lowest)

max. 45

Stream Conditions Total:

Total IHAS Score: %

CHAPTER 4

DESCRIPTION OF THE STUDY AREA

4.1 INTRODUCTION

The Luvuvhu River in the Northern Province of South Africa represents a river significant both from a human and ecological perspective (Kleynhans, 1996). The Luvuvhu River catchment is one of the main sources of water for domestic and agricultural purposes in the Northern Province. A part of the river also flows through the Kruger National Park, where it provides water to wildlife in the dry northern parts of the reserve.

The mainstream of the Luvuvhu has a length of approximately 200 km. The catchment of the Luvuvhu River, excluding the Mutale tributary, covers an area of 3 470 km² (3 568 km² including the Mutale), and in 1985 had a human population of 270 500 (Kleynhans, 1996). The population density in 1996 was 85 people/km². The catchment originates in the Soutpansberg mountain range and has several main tributaries namely the Mutale, Dzindi, Mutshindudi, Latonyanda, and Mbwedi Rivers. The Luvuvhu River finally flows into the Limpopo River at the South Africa/Zimbabwe/Mozambique border.

Rainfall, which occurs in summer, ranges from 2 068 mm/y in a relatively small area on the northwestern slopes of the Soutpansberg mountain range, to 440 mm/y near the confluence with the Limpopo River. The natural vegetation in the catchment includes Inland tropical forest, Tropical bush and Savanna.

According to Claassen (1996), the geology in the catchment consists of a variety of geological units with the most important ones being: Baberton, Murchison, Giyani, Beitbridge, Suurberg, Drakensberg, Lebombo, Waterberg, Soutpansberg, Orange River alluvium, Sand, Calcrete, Meinhardskraal granite and Sand River gnies. The four main soil types present in the catchment are Glenrosa, Hutton, Mispah and Shortland.

4.2 GENERAL HYDROLOGY OF THE LUVUVHU RIVER

The hydrological features of a catchment reflect the integrated effects of climate, topography, soils, veld types and land use on the distribution of surface water in time and space (Heath, 1999).

The small high altitude area (>1 200 m a.m.s.l.) contributes most of the Luvuvhu River's runoff through the contributions of perennial tributaries such as the Mutshindudi, Dzindi and the Latonyanda Rivers. These three rivers supply respectively 22.7, 10.9, and 11.9 percent of the total virgin runoff of the Luvuvhu (570 million m^3/y) at its confluence with the Limpopo River at 232 m a.m.s.l. (Kleynhans, 1996).

The construction of physical structures such as dams directly alter hydrology by constraining the flow of the river. The decrease in water flow may reduce the natural diversity and abundance of a wide range of fishes and invertebrates. Further, flow stabilization below water supply reservoirs results in artificially constant environments eliminating species adapted to natural dynamics (EPA, 1998). Water withdrawals for agricultural uses also affect even the most tolerant species, where the minimum flow isn't provided.

Several water supply impoundments, as well as flow gauging weirs, for the provision of water for agriculture, are found in the Luvuvhu River. Details of four dams in the Luvuvhu River are shown in Table 4.1, for 1994.

Table 4.1. Physical characteristics of four dams in the Luvuvhu River catchment area.

Dam	Capacity (million m³/a)	Mean Annual Runoff (million m³/a) - Net
Vondo	5.3	30.8
Albasini	25.6	14.4
Tshakhuma	2.1	7.0
Mambedi	7.0	2.9
Total	40.0	55.1

4.3 LAND USE IN THE STUDY AREA

The Luvuvhu River is influenced by three completely contrasting landscape practices. The first of these land uses is in the upper reach of the river, where modern first world farming is practiced. In the lower reach subsistence farming becomes the dominant land use. The final part of the river is then situated in a National Park. These contrasting practices lead to the river being exposed to entirely different impacts.

4.3.1 Agriculture

A large amount of water is abstracted in especially the upper reaches of the Luvuvhu River for agricultural use. A total of 15.4 million m^3/y is allocated to irrigate 1 845 ha from the Albasini Dam and from weirs on the Luvuvhu and the Latonyanda Rivers by an extensive system of interlinking canals. The virgin runoff at the downstream end of the upper reach (approximately where the Latonyanda flows into the Luvuvhu mainstream) was reduced from 134.61 million m^3/y to 86.20 million m^3/y in 1987 (Kleynhans, 1996). This reduction was primarily due to water abstraction for agricultural purposes.

Agricultural chemicals, overgrazing, removal of riparian vegetation, erosion, and sedimentation are also some of the environmental problems that are associated with first and third world agricultural activities present in the catchment.

4.3.2 Forestry

Forestry covers an area of 14 600 ha in the river's upper reaches (Claassen, 1996), and reduces runoff.

4.3.3 Nature conservation

A section of the Luvuvhu River lies within the Kruger National Park. Although this section of the river is mostly undisturbed, the landscape practices in the reaches leading up to the Kruger National Park do have a definite affect, e.g. water abstraction, flow modification, erosion, etc. (Kleynhans, 1996).

4.3.4 Industry

Small industries dominate this land use type, with a roller mill, a brewery, saw mills, and various other small industries.

4.4 SELECTION OF SAMPLING LOCALITIES

Sampling localities were chosen at historical fish monitoring sites.

Refer to Figures 4.1 and 4.2 for the locality maps of the study area. The positions of the sampling localities in the Luvuvhu River catchment are provided in Figure 4.3. All relevant information concerning the sampling localities is presented in Table 4.2.

Figure 4.1: Location of South Africa in relation to Africa

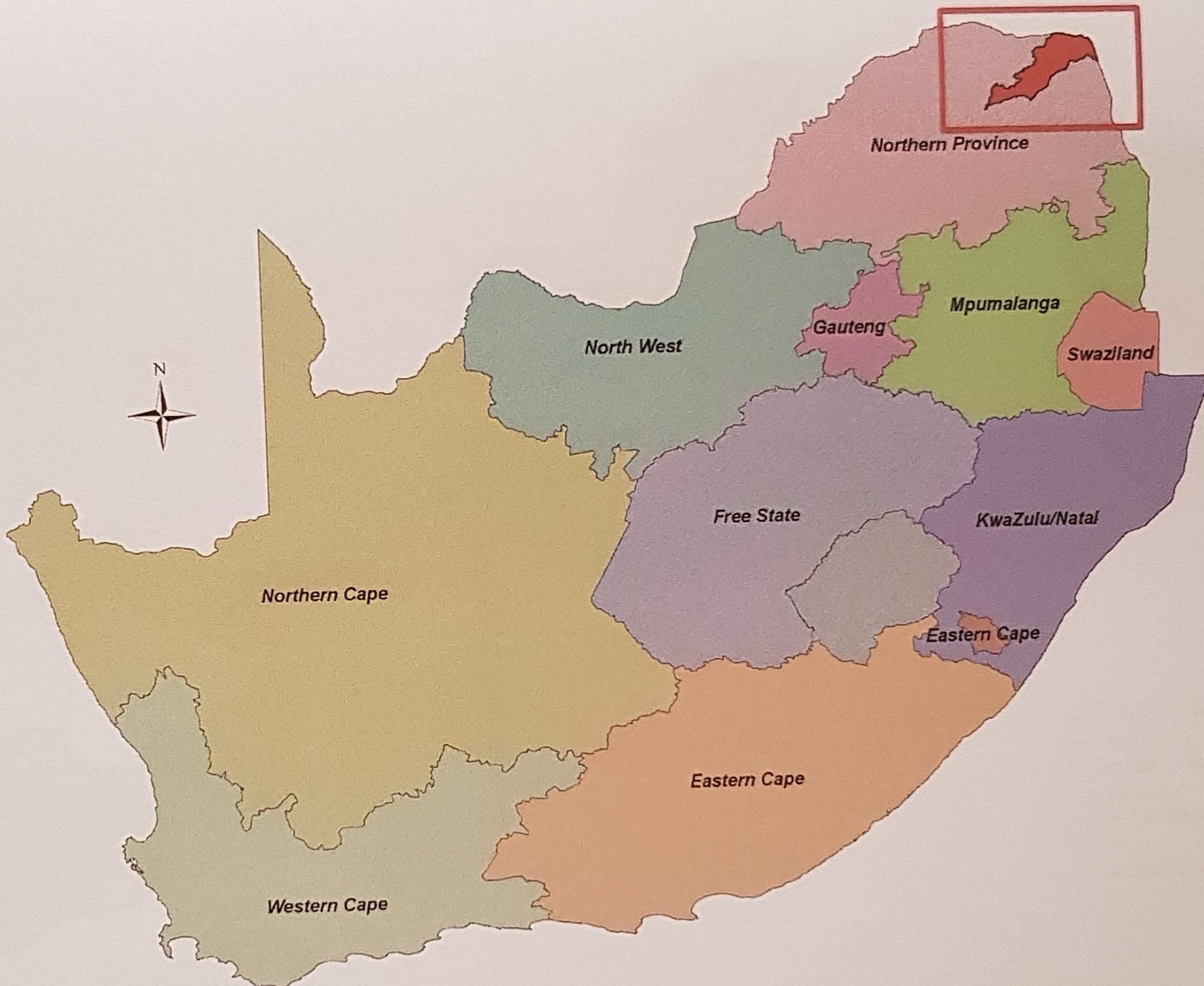

Figure 4.2: Location of the Luvuvhu River catchment in relation to South Africa

Figure 4.3: Sampling localities in the Luvuvhu River catchment

Table 4.2: Sampling localities in the Luvuvhu River catchment.

DATE	SAMPLING LOCALITY NO.	SAMPLING LOCALITY	COORDINATES S		COORDINATES E	
			Degrees	Minutes	Degrees	Minutes
19/08/99	1	Top bridge	22	59.35	30	19.07
19/08/99	2	Forest track below water fall	22	58.57	30	20.05
18/08/99	3	Bridge by crocodile ventures	23	0.38	30	28.41
19/08/99	4	Botha's farm bridge	23	3.08	30	14.07
18/08/99	5	Cabbage farm	23	4.47	30	19.27
16/08/99	6	Above Albasini	23	4.08	30	4.05
16/08/99	7	Shefeera	23	0.02	30	0.05
17/08/99	8	Beja bridge	23	5.51	30	4.03
17/08/99	9	Valdezia	23	5.10	30	10.28
30/08/99	10	Robert's farm	23	6.18	30	20.45
17/08/99	11	G.weir below Luvuvhu	23	6.51	30	23.26
18/08/99	12	Hasani crossing	23	5.04	30	28.16
01/11/99	13	Nandoni	22	58.29	30	36.10
18/11/99	14	Malamulele pump-works	22	57.15	30	38.94
04/11/99	15	Tshifundi	22	50.57	30	45.09
22/09/99	16a	Botsileni channel 1	22	47.25	30	50.91
22/09/99	16b	Botsileni channel 2	22	47.25	30	50.91
22/09/99	16c	Botsileni channel 3	22	47.25	30	50.91
22/09/99	16d	Botsileni channel 4	22	47.25	30	50.91
01/11/99	17	Mhinga pump	22	45.18	30	53.35
21/10/99	18	Lambani	22	44.19	30	52.93
18/10/99	19	Dongadziva (KNP 1)	22	42.53	30	53.35
18/10/99	20	Shidzivane (IFR 2) (KNP 2)	23	40.00	30	57.50
20/10/99	21	Madzaringwa (KNP)	22	29.90	31	3.57
19/10/99	22	Mutale bend (KNP)	22	26.67	31	4.56
19/10/99	23	Mangala (KNP)	22	25.62	31	10.46
19/10/99	24	Bobomene (KNP)	22	25.50	31	12.50
18/11/99	25	Mphaphula cycad reserve	22	48.62	30	38.87
03/11/99	26	Damani Dam pump	22	50.58	30	31.10
04/11/99	27	Bridge above Mutsh. Confluence	20	50.09	30	39.43
31/08/99	28	Phiphidi falls	22	56.00	30	23.00
31/08/99	29	Phiphidi hydro-bridge	22	56.21	30	24.04
31/08/99	30	Tshivulani	22	54.54	30	29.18
31/08/99	31	School turn and water fall	22	53.17	30	35.21
04/11/99	32	Malavhuve bridge	22	51.40	30	38.37
21/09/99	33	New guaging weir	22	51.20	30	41.13
01/09/99	34a	Tshiombedi - Above falls	22	45.43	30	28.50
01/09/99	34b	Tshiombedi - Low bridge	22	45.43	30	28.50
03/11/99	35	Second bridge	22	43.10	30	39.03
02/09/99	36a	Tshirova riffle	22	48.55	30	23.47
02/09/99	36b	Tshirova junction pool	22	48.55	30	23.47
01/09/99	37	Narrow roadside	22	48.51	30	25.86
01/09/99	38	Whboneni School bridge	22	47.34	30	26.56
02/09/99	39	Samb. Bridge	22	42.04	30	38.34
03/11/99	40	Tshikundamalema	22	40.28	30	42.09
02/11/99	41	Guyuni	22	35.16	30	48.32
02/11/99	42	Tshikondeni bridge	22	28.44	30	52.83

4.5 SAMPLING FREQUENCY

The 42 localities in Table 4.2 were sampled over a three-month period during spring and early summer. The monitoring plan for the Luvuvhu River tied in with the river research programme for the Kruger National Park.

4.6 REFERENCES

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CHAPTER 5

AQUATIC MACROINVERTEBRATE AND HABITAT ASSESSMENT

5.1 INTRODUCTION

In general, high impact anthropogenic activities (e.g., industry and mining) are scarce in the Luvuvhu catchment area. The river is mostly free from serious chemical pollution and can be considered as having high biological integrity.

In the western half of the catchment (before Thohoyandou) land use is dominated by first world agriculture. Citrus, mango, and avocado farms are common. Cattle-farming, although sparse, is also present. In the more upper reaches of the Luvuvhu River forestry activities are found. All these landscape practices cause several environmental problems. According to Ortolano (1984), the following residuals are commonly associated with farm and forestry runoff:

- Agriculture: croplands - sediments, pesticides, compounds of phosphorus and nitrogen, and total dissolved solids.
- Agriculture: animal feedlots - biodegradable organic matter, pathogenic organisms, and compounds of phosphorus and nitrogen.
- Commercial forests - sediments, pesticides, water temperature

Agricultural impacts are mainly caused by irrigation activities. Withdrawals of water can eliminate streams, reduce habitats, or impoverish vegetation by lowering groundwater levels (USEPA, 1998). According to Kleynhans (1996), the reduction of water in the Luvuvhu River due to abstraction in the upper tributaries for agricultural purposes is the most prominent modification to the habitat integrity. The water quantity is reduced the most during the dry winter months.

The waters in the areas surrounding the upper reaches are also at risk of being contaminated by pesticides used in pest control operations, as agriculture and forestry are the activities that are considered to be the principle users of pesticides (Nriagu and Lakshminarayana, 1989). Pesticides include a wide range of toxic chemicals. Well-known examples include organochlorine insecticides, and organophosphrous compounds. The tolerance levels of a particular species of stream macroinvertebrate may differ widely according to the nature of the pesticide (Muirhead-Thomson, 1987). High concentrations may lead to mortality. Less severe reactions include behavioral responses, e.g. immobilization, escape reactions, case-leaving by caddis larvae, and burrow leaving by chironomids (Muirhead-Thomson, 1987). Drift reactions have also been observed with different species (e.g., *Simuliidae*), where the term drift refers to the downstream transportation of invertebrates by stream currents (Mokgalong, 1981).

Other sources of agricultural pollution include the wastes of animals (cattle farms), and runoff of inorganic fertilizers. Both cause an increase in the levels of phosphates and nitrates, leading to eutrophication. According to Freedman (1989), eutrophication refers to the process by which an aquatic ecosystem increases in productivity as a result of an increase in the rate of nutrient input. The organic and inorganic matter serves as food for decomposers such as *Simuliidae*, *Chironomidae* and *Oligochaeta*. Larvae of stoneflies, caddisflies and mayflies, which respire with gills or by direct cuticular exchange are particularly susceptible to the resulting decrease in oxygen levels (Dallas, 1995). There is thus a shift in the community structure to those taxa that are more tolerant and suited for these conditions.

Forestry decreases in importance from west to east in the catchment. It is well known that afforestation activities reduce runoff. In the Crocodile River catchment afforestation is estimated to have reduced runoff by approximately 20% (Heath, 1999). During rainfall large quantities of suspended sediments, dissolved salts, and nutrients are also added to the receiving waters.

As the Luvuvhu River progresses into the eastern part of the catchment, third world land use becomes prominent. Informal settlements are abundant with numerous prevailing impacts, such as subsistence farming, washing, grazing (cattle and goats), trampling, tree cutting, and tree felling.

Once again, the farming activities may lead to eutrophication as a result of nutrients from the farming activities reaching the streams and river through runoff. Irrigation and diversions of the water also leads to a reduction in the water quantity.

Various weirs and dams have been constructed in the Luvuvhu River to supply water to the surrounding settlements. The impoundment of water may lead to a reduction in habitat available to aquatic fauna and may obstruct movement of aquatic fauna, and influences water quality and sediment transport (DWAF, 2000).

A common phenomenon in the Luvuvhu River is the constant presence of woman washing clothes in the riffles. The soap used is viewed as a man-made surfactant (surface-active agents). They reduce the rate of re-aeration of water and thus reduce the amount of oxygen available to the biota (Dallas, 1995). Although the effects of non-biodegradable detergents are much more severe than with biodegradable detergents, it is not known what type was mostly used.

Natural riparian vegetation, which forms an integral part of any ecosystem, plays an important role in river bank stabilization (O'Keeffe, 1986). Unfortunately, it is frequently destroyed for various reasons, including allowing the planting of crops, facilitating the movement of people and livestock, and extending grazing areas. All these activities, which were present in the Luvuvhu catchment, will lead to an increase in the suspended solids in the river water.

Along the Luvuvhu River the destruction of the riparian vegetation has also cleared the way for exotic plant species, e.g., *Lantana camara*. According to the USEPA (1998), successful riparian management should include the following general principles: (1) fence of herd stock out of riparian areas, (2) control the timing of grazing to keep the stock off stream banks that are most vulnerable to erosion and to coincide with the physiological needs of plants, (3) provide more rest to the grazing cycle to increase plant vigor or encourage more desirable species, and (4) limit grazing intensity. Unfortunately, the implementation of these principles would be an almost impossible task in the third world conditions of the Luvuvhu catchment. A realistic approach should thus be considered in the management of this and similar areas.

In densely populated villages there is a large-scale removal of natural vegetation for fuel. This causes large areas to be exposed. The "trampling effects" of humans and stock, especially cattle, tend to decrease the permeability of the upper soil layers. This, in turn, causes a greater proportion of any rainfall to run off the surface of the catchment, often transporting large quantities of soil and other particulate and dissolved material (Heath, 1999). This then causes an increase in the amount of suspended solids in the river water, which has various effects on the river biota and habitat. According to Hellawell (1986), the deposition of solids will cause a lowering in the benthic community diversity through the disappearance or marked reduction in biomass and numbers of certain sensitive species. Further, there is a reduction in light penetration as a consequence of increased turbidity, mechanical effects on organisms (e.g., clogging of gills, interfering with the feeding of filter feeders), and a modification in the nature of the habitat through the change in the character of the substratum. This will lead to an alteration in the benthic community structure, where the silt-tolerant organisms, that are able to cope with the change in habitat and surrounding conditions, will dominate.

Almost every rural community lacked piped water supplies and water-borne sanitation systems. The human wastes add organic and inorganic wastes to the water. As discussed, the process leads to eutrophication, which induces alterations to the invertebrate community structure.

Another problem is the constant crossing of streams by cattle and humans. When streams are waded through, especially at riffle areas, the microhabitats are disrupted and invertebrates drift downstream. This effect becomes more severe at places that are crossed on a regular basis.

In the Kruger National Park the Luvuvhu River is exposed to minimal stress. Unfortunately, upstream abstraction has a very high negative impact on this final reach of the river, considering that a large volume of water is required to saturate the alluvial floodplain before surface flow can occur (Kleynhans, 1996). Dams and reservoirs, as well as elevated abstraction of water during dry winter months, cause flow modifications. The duration of low and no flow conditions is considerably increased by upstream modifications.

It is a generally accepted view that sediment acts as a sink for trace metals (Dallinger and Rainbow, 1993). Organic and other particulate material will also, eventually, settle out of the water column and rest in the sediment. Sediment can hence be seen as the repository of most contaminants (Jones *et al*, 1999). Contaminants incorporated into the sediments are also generally more persistent and less mobile than those in the overlying waters. Contaminated sediments thus pose a severe threat to the benthic biota, as they mostly inhabit the bottom surfaces of streams and rivers and are thus in constant contact with the underwater sediments. For example, the input of inorganic particulate materials may affect benthic organisms from both habitat destruction and physical damage to the biota (Worf, 1980). Thus, higher levels of sedimentation can affect aquatic insects by altering biochemical conditions, food resources, respiratory diffusion gradients, and habitat space (Williams and Feltmate, 1992). The sediment particle sizes and contaminant content varies according to the site locality, river flow, and upstream land-use.

At weirs particles settle due to slower flow caused by the weir. The build up of silt just behind the weir may cause nutrient-poor water to flow into the downstream river. In the Luvuvhu River the localities situated just after weirs would hence have been adversely affected. The only noticeable negative influence caused by gauging weirs was restricted to a reduction of flow.

The nature of the sediment could also influence what taxa are found. For example, in the Caledon River catchment, *Leptophlebiidae* were more frequently found in the Lesotho Highlands (sand of dolomitic origin) than in the Lesotho Lowlands (predominately silt originating from Karoo Sequence Formations) and the opposite was true for *Tricorythiidae* (Chutter, 1998).

Finally, the Luvuvhu River provides an ample amount of diverse habitats to all forms of aquatic biota. The sampling localities were chosen where the main biotopes were present, namely the stones in current, stones out of current, and marginal vegetation. These are also then the habitats that are preferred by most benthic macroinvertebrates. According to Dallas (1995), taxa percentages decrease in the stones in current, marginal vegetation, stones out of current, aquatic/instream vegetation and sand respectively.

5.2 MATERIALS AND METHODS

SASS4 and IHAS were performed at all the localities shown in Table 4.2.

5.2.1 Benthic Macroinvertebrates (SASS4)

Field sampling was carried out according to the following instructions given by DWAF (2000) (included in SASS4 field record sheet, Table 3.2):

- SIC: riffle and run, sample for 2 min if all kickable, otherwise for a maximum of 5 min.
- SOOC: backwater and pool, kick about 1 m².
- Marginal vegetation: back and forward sweep - 2 m.
- Instream/aquatic vegetation.
- Gravel, sand, and mud: stir with feet and sweep net over disturbed area for 0.5 minute.

The net contents of each of the above (where available) was placed in a tray and the leaves and twigs were then examined and removed. The taxa present were then determined, taking no more than 15 min or 5 min since the last taxon was identified. The abundances were then estimated using the following scale:

- 1: 1
- A: 2-10
- B: 10-100
- C: 100-1000
- D: >1000

After identification, each score sheet was completed and the sample score, number of taxa, and Average Score Per Taxon (ASPT) were calculated.

The SASS4 scores were then interpreted according to the following guidelines proposed by Chutter (1998), where surface waters are not naturally acid (pH >6):

SASS4 > 100, ASPT > 6	water quality natural, habitat diversity high
SASS4 < 100, ASPT > 6	water quality natural, habitat diversity reduced
SASS4 > 100, ASPT < 6	borderline case between water quality natural and some deterioration in water quality, interpretation should be based on the extent by which SASS4 exceeds 100 and ASPT is <6
SASS4 50-100, ASPT < 6	some deterioration in water quality
SASS4 < 50, ASPT variable	major deterioration in water quality

5.2.2 Habitat (IHAS)

In the first section of the IHAS score sheet, each type of sampling habitat was allocated a value, according to characteristics of the habitats that were sampled, availability of biotopes and the condition of the stream. All the scores at each subsection were added to produce an individual total score for 'stones in current', 'vegetation', and 'other habitat'. Adjustment numbers were calculated by subtracting these total scores of each subsection from the maximum scores (20, 15, and 20 for each subsection respectively). A total adjustment score was then obtained and added to the SASS4 score of each locality.

In the second section, the physical characteristics of the stream were scored.

The IHAS scores were interpreted according to McMillan (1998), where it is presently thought that a total score of over 75% represents good habitat conditions and over 65% indicates adequate habitat conditions.

5.2.3 Water Quality

In situ measurement of the water quality was performed by utilising standard measurement techniques and apparatus (i.e. hand-held field instruments). The following chemo-physical variables were measured:

1. Temperature
2. Conductivity
3. pH

5.3 RESULTS

5.3.1 SASS4

The results from the completed SASS4 score sheets as presented in Appendix 5A, Table 5A.1, include the sample score, number of taxa, and Average Score Per Taxon (ASPT) for each locality. The different invertebrate families that were sampled and their abundances are provided in Appendix 5A, Table 5A.2a-c.

Table 5.1 contains the evaluation of the SASS4 scores and the ASPTs, according to Chutter's interpretation guidelines. Six of the sampling localities deviate from natural water quality and high habitat diversity, namely Valdezia (sampling locality 9), Botsileni channel 3 (sampling locality 16c), Botsileni channel 4 (sampling locality 16d), Damani Dam pump (sampling locality 26), Second bridge (sampling locality 35), and Tshirova junction pool (sampling locality 36b).

5.3.2 IHAS

The IHAS score sheet was completed at each sampling locality (refer to Appendix 5A, Table 5A.3). In total, only five localities consisted of inadequate habitat conditions. These were at Bridge by crocodile ventures (locality 3), Cabbage farm (locality 5), Shefeera (locality 7), Beja bridge (locality 8), and Valdezia (locality 9).

The necessary adjustments were made to the SASS4 scores, and are indicated in Appendix 5A, Table 5A.4.

5.3.3 Water Quality

The water quality variables that were collected are shown in Appendix 5A, Table 5A.5. Unfortunately, the recording of these variables was incomplete, thus creating a data gap.

5.4 DISCUSSION

5.4.1 SASS4 Scores and ASPTs

Of the six sampling localities (9, 16c, 16d, 26, 35, 36b) that deviate from natural water quality and high habitat diversity, two localities (9 and 16c) are borderline cases between water quality natural and some deterioration in water quality.

At Valdezia (sampling locality 9) the ASPT was 5.51 and the SASS4 score was 117. This locality was situated below a gauging weir, where there was little flow, causing an absence of the biotope "stones in current". In riffle/run areas high taxa richness and abundance occur (Rosenberg and Resh, 1993). According to O'Keeffe (1986), the fauna that inhabits the stones in current has generally been shown to be the most responsive to water quality and environmental change. An example of a high scoring taxon that inhabits riffle areas, and that was not found, is *Perlidae* (sensitivity score of 12). It was thus a reduction in habitat diversity and not the water quality that was to blame for the reduced ASPT.

Botsileni (sampling locality 16) was a multi-channel system caused by a series of islands, giving the main channel a braided appearance. It was decided to sample the four largest channels in order to compare their species composition and to get a true reflection of the biological richness. Channels 1 and 2 (sampling locality 16a and 16b respectively) were the main channels, based on their depth, width and water flow rate. High SASS4 scores and ASPTs were recorded for these two channels. For channel 3

Table 5.1: Interpretation of SASS4 scores and ASPTs.

Locality no.	SASS4 scores	ASPT	Water Quality	Habitat Diversity
1	184	8	natural	high
2	160	6.95	natural	high
3	133	8.31	natural	high
4	123	6.83	natural	high
5	132	6.94	natural	high
6	230	7.66	natural	high
7	165	7.17	natural	high
8	137	6.85	natural	high
9	117	5.51	borderline case	
10	146	8.1	natural	high
11	169	7.04	natural	high
12	167	6.95	natural	high
13	107	6.68	natural	high
14	129	6.14	natural	high
15	169	6.76	natural	high
16a	179	6.88	natural	high
b	173	6.17	natural	high
c	101	5	borderline case	
d	87	4.83	some deterioration	
17	173	6.92	natural	high
18	153	6.95	natural	high
19	180	6.43	natural	high
20	174	6.21	natural	high
21	176	6.76	natural	high
22	199	6.86	natural	high
23	156	7.09	natural	high
24	203	7.52	natural	high

Table 5.1: (continued)

Locality no.	SASS4 scores	ASPT	Water Quality	Habitat Diversity
25	140	7.37	natural	high
26	77	5.13	some deterioration	high
27	137	6.85	natural	high
28	115	6.38	natural	high
29	144	7.2	natural	high
30	153	7.65	natural	high
31	117	6.5	natural	high
32	146	6.34	natural	high
33	184	6.34	natural	high
34a	140	7.78	natural	high
b	120	6.67	natural	high
35	99	5.5	some deterioration	high
36a	209	8.36	natural	reduced
b	62	6.8	natural	high
37	189	7.27	natural	high
38	149	6.77	natural	high
39	138	6.9	natural	high
40	127	7.47	natural	high
41	180	7.83	natural	high
42	134	7.44	natural	high

(sampling locality 16c), the third largest channel, there were no riffles and thus an absence of high scoring taxa that are usually found in this specific biotope (e.g., *Perlidae* and *Leptophlebiidae*). Hence the lower SASS4 score (101) and low ASPT (5.0). Channel 4 (sampling locality 16d) was the smallest channel, and was mostly backwater of channel 1. It contained the least amount of water and no taxa of *Trichoptera* were sampled. This caused channel 4 to have the lowest SASS4 score and ASPT for Botsileni. Once again, as the high SASS4 scores and ASPTs in the main channels proved, there was no deterioration in water quality at this locality, but a reduction in habitat diversity in the smaller channels.

Two localities indicated some deterioration in water quality, namely Damani Dam pump (sampling locality 26) and Second bridge (sampling locality 35).

Damani Dam pump (locality 26, Figure 4.3) is located below a gauging weir. The SASS4 score (77), ASPT (5.13), and number of families scored (15) were low. The area surrounding the locality was coloured red-brown, because of the rust deposit from the weir. This colour was also apparent on the rocks in the water and on the leaves of the marginal vegetation. Most of the stones and rocks were also covered with fine debris and deposited sand. In comparison with other localities, the lack of *Ephemeroptera* taxa would suggest that this Order of insects suffered the most. These reductions can be attributed to the presence of the iron oxide and the loss of habitat. Visually, this was certainly also the worst locality due to the discoloring.

A similar situation was discovered at Second bridge (sampling locality 35) in the Sambandou River, where very few *Ephemeroptera* were sampled - only two species of *Baetidae*. This insect group is sensitive to water quality (Loch *et al*, 1996). They inhabit the stones and rocks in riffles. The stones and rocks were however unmovable due to cementation as a result of silt deposits, which caused a loss of interstitial space between the stones. These insects could thus not reach the bottom of the rubble to inhabit these affected areas.

The scores for the Tshirova riffle at sampling locality 36a (SASS4=209, ASPT=8.36), were of the highest obtained at all the sampling localities. The riffle flowed into a large pool, and because of its size it was decided to sample it on a different score sheet as locality 36b. Only the stones out of current, vegetation, and bedrock were present and sampled, and hence the lowest SASS4 score for all the sampling localities was attained at the Tshirova junction pool (locality 36b). This low score (62) is however not a true reflection of the high biological quality of this locality.

Considering the high average (149) of the SASS4 scores obtained there was decided to look at sampling localities where scores below 130 were recorded, to determine the reasons for deviations. Localities with these lower scores, that haven't been discussed yet, include 4, 13, 28, 31, 34b, and 40. It should be remembered that these SASS4 scores were still all above 100, and indicate good overall quality.

At Botha's farm bridge (sampling locality 4), the surrounding riparian vegetation was felled to make way for pine plantations. The increase in sedimentation caused almost 50% of the rocks to be unmoveable. This would have diminished the microhabitats that are inhabited by invertebrates.

Nandoni's (sampling locality 13) low SASS4 score can partly be attributed to the lack of stones out of current and the abundance of mud. There were also no cased caddis larvae present, which normally contribute to high SASS4 scores. No vegetation was sampled at Phiphidi falls (sampling locality. 28), which explains the less than average score that was obtained for this locality.

At School turn and water fall (sampling locality 31), only one taxon representing *Odonata* was recorded. Throughout the localities sampled, *Odonata* taxa were well represented, and mostly score high on the score sheet. It is uncertain what caused the shortage of these taxa leading to a reduced SASS4 score.

Besides sampling locality 16 and 36, Tshiombedi (sampling locality 34) was also sampled more than once. Part of the sampling locality (34a) was located above a small waterfall and scored above average. The second part (locality 34b) was below the hill, just after a pool. There was a large amount of litter at this part, in the form of maize, which was being extensively washed further up in the river. The maize was especially concentrated around the marginal vegetation. This could be to blame for the absence of any *Hemiptera* and the lower SASS4 score, possibly due to high BOD and resultant lower dissolved oxygen levels.

Most of the rubble at Tshikundamalema (sampling locality 40) was unmovable due to cementation, which was caused by sand deposition. This lead to less *Ephemeroptera* being present, and a lower SASS4 score.

In general, the least sampled biotopes were aquatic vegetation, mud, gravel, and bedrock. Fortunately, the SASS4 scores weren't greatly influenced by the absence of these biotopes, considering that almost all the invertebrate families prefer the remaining available habitats, namely the stones in current, stones out of current and marginal vegetation.

Natural changes in environmental factors (e.g. flow, water temperature, dissolved oxygen and food sources) along the longitudinal profile of river systems exert a direct control on the population dynamics of aquatic organisms, resulting in characteristic biological communities and zones (Dallas, 1997). There were, however, very few patterns in the species compositions in the different sub-regions (i.e. mountain stream, foothill, transitional, and lowland river) of the Luvuvhu River. Most taxa had a constant distribution (refer to Appendix 5A, Table 5A.2a-c).

A few families showed minor changes as the Luvuvhu mainstream (from sampling locality 7 to 24) progressed downstream. The following families increased in presence and abundance: *Oligochaeta*, *Planorbidae*, and *Sphaeridae*. In a downstream direction, sandbanks become more common in the Luvuvhu River. These habitat conditions suit *Sphaeridae* and as filterfeeders these bivalves also prefer the increase inorganic matter from the farms, rural settlements, cattle, etc., as do the *Oligochaeta*.
It is also expected that changes should occur in scores resulting from longitudinal differences in river systems. Dallas (1995) and Chutter (1998) found that, generally, SASS4 scores and ASPTs decline in a downstream direction. No clear trend was however evident from the SASS4 scores and ASPTs obtained for the different sub-regions in the Luvuvhu River catchment, except in the Mutale River (Appendix 5A, Table 5A.1). Here there was a decrease in the SASS4 scores from the mountain, foothill, and transitional areas down to the lowland areas (sampling locality 36 - 40).

The relationship between the different feeding types (i.e. shredders, collectors, scrapers, and predators) showed no particular pattern, and there weren't any shifts in the dominant species in the community, e.g. from shredders in the upper reaches to collectors in the lower areas.

At the sampling localities situated in the Kruger National Park (KNP) high SASS4 scores (average = 181.33) and ASPTs (average = 6.81) were recorded. At sampling locality 24 five types of cased caddis larvae were found, which was only one in three localities in the whole catchment where this was recorded. At four localities in the KNP the scarce *Philopotamidae* was identified. It can thus be concluded that the sampling localities in the KNP are free of serious stress, and aren't severely impacted by the upstream landscape activities.

Based on Table 5A.2a-c in Appendix 5A, there were no incidents where there was a shift in the community structure to taxa that are more tolerant and suited to eutrophic conditions. The reason for this may be that no localities were overly enriched by nutrients. This was verified by the overall low algal and fungal presence, where the algae was mostly isolated and attached to rocks. The localities also mostly had rapidly running water, which, according to Kupchella and Hyland (1993), decreases the oxygen removing effect of the decomposers.

Although the use of surfactants for washing clothes is a common occurrence in the river, it might be assumed that the soap is of such a nature (i.e. biodegradable) that the invertebrates aren't severely affected, as high SASS4 scores persisted in these areas.

5.4.2 IHAS

IHAS attempts to account for the variability in the amount and quality of habitats or biotopes available for habitation by aquatic biota (DWAF, 2000).

According to McMillan (1998), it is presently thought that a total score of over 75% represents good habitat conditions, and over 65% indicates adequate habitat conditions.

Taking the above mentioned into account, sampling locality 1, 4, 11, 12, 15, 16 (channel 1 and 2), 19, 25, 26, 27, 34, 35, 36 and 37 can be considered as having good habitat conditions. Further, sampling localities 2, 6, 10, 13, 14, 17, 18, 20, 21, 22, 23, 24, 28, 29, 30, 31, 32, 33, 38, 39, 40, 41 and 42 indicate adequate habitat conditions (refer to Appendix 5A, Table 5A.3).

In total, only five localities (3, 5, 7, 8, and 9) consisted of inadequate habitat conditions. According to the SASS4 interpretation guidelines (Chutter, 1998), locality 3, 5, 7 and 8 indicated high habitat diversity. Although there seems to be a clash of habitat interpretation, this is not the case. The scores for the Sampling Habitat (refer to Table 3.3 for IHAS version 2 score sheet) were high enough to confirm that habitat diversity was sufficient, which corresponds to the SASS4 derived interpretations. The Stream Condition on the IHAS version 2 score sheet is mostly to blame for low total IHAS scores, where erosion and a low percentage of riverbank covering were the primary causes. The invertebrates were not sufficiently adversely affected to have caused reductions the SASS4 scores. At sampling locality 9 both SASS4 and IHAS interpretations show poor sampling habitat diversity. This was due to the absence of the biotope stones in current (SIC).

The highest adjustment values were generally calculated at the Other Habitat/General section of the IHAS score sheet. This can be attributed to the general absence of mud, gravel, and bedrock.

When analyzing the IHAS scores it is important to remember that a high total IHAS score doesn't necessarily indicate that both Sampling Habitat and Stream Condition are of high quality. For example, at sampling locality 36 (Tshirova junction pool and riffle) the Sampling Habitat had a score of 50, whereas the Stream Condition scored at 35. The same situation can occur with the Stream Condition scoring a lot higher than the Sampling Habitat.

The importance of habitat diversity has already been stressed, as the absence of a major habitat requirement may substantially reduce the SASS4 score (McMillan, 1998). The SASS4 scores were adjusted using the adjustment values generated with the IHAS scoring sheet (Appendix 5A, Table 5A.4). This was performed to modify the SASS4 scores to reflect the 'ideal' habitat conditions set for all rivers and streams. All the SASS4 scores were increased. This modification tries to account for different sampling habitats (McMillan, 1998). For example, after modifications sampling locality 7 and 41 have the same SASS4 score (namely 194). Before the adjustment however, locality 7 had a SASS4 score of 165 and 41 had a SASS4 score of 180. At locality 7 there was no vegetation sampled and a much higher adjustment value assigned to the SASS4 score. IHAS thus compensates for the lack of sampling habitat.

5.4.3 Selected Site-specific Physico-chemical Parameters

Unfortunately, not enough water quality data was obtained to deliver a noteworthy contribution to the evaluation of the SASS4 results. Where low SASS4 scores and ASPTs were obtained, conclusions concerning these results were mainly based on the habitats sampled and the surrounding impacts (e.g., gauging weirs).

According to the interpretation guidelines mentioned above, there is a decrease in water quality when the ASPT is lower than 6 (recorded at sampling locality 9, 26, and 35). These lower scores can be ascribed to the decrease in the higher scoring taxa that are intolerant to pollution (e.g., *Perlidae* and *Leptophlebiidae*). The absence of adequate water quality data, however, made it difficult to find fault with the water quality at these localities. Low scores are thus, once again, primarily blamed on unfavorable habitat conditions.

Dallas *et al* (1999) set out to deduce ranges of different water quality variables, for different taxa. Table 5.2 provides an example of the ranges of conductivity and pH where three invertebrate families, with different tolerances to water quality impairment, are commonly found.

For this study, the average pH for the sampled localities is 7.7, with a standard deviation of 0.24. With pH values so closely together it is difficult to find similarities between taxa in Table 5.2 and those in Table 5A.2a-c in Appendix 5A.

The conductivity showed more of a variance. There was an increase in conductivity along the length of the catchment. This increase coincides with the more numerous anthropogenic influences as the river progresses, e.g. agricultural runoff and rural settlement activities. A more in-depth study needs to be undertaken to determine the preferences of certain taxa to conductivity ranges. In this study no clear conclusions on this topic can be reached.

Table 5.2: The recorded ranges of conductivity and pH for *Ephemerellidae* (SASS score = 15), *Heptageniidae* (SASS score = 10) and *Chironomidae* (SASS score = 2) (Dallas *et al*, 1999). SD = Standard Deviation, Min = Minimum, Max = Maximum.

Chemical variable	Family	Average	SD	Min	Max	Range
Conductivity	*Ephemerellidae*	3.8	2.7	1.2	16.1	14.9
Conductivity	*Heptageniidae*	14.9	13.2	1.7	54.1	52.4
Conductivity	*Chironomidae*	42.6	49.7	2.1	227.0	224.9
pH	*Ephemerellidae*	6.18	0.73	4.4	7.60	3.20
pH	*Heptageniidae*	7.19	0.80	5.3	8.58	3.28
pH	*Chironomidae*	7.33	0.86	4.8	8.90	4.10

As far as temperature is concerned, cooler temperatures were recorded in the higher upper catchment with an increase in temperature in the lower catchment. According to Worf (1980), only temperatures in excess of 30°C may result in substantial changes in the benthic fauna. Fortunately, these elevated temperatures weren't encountered, with the temperature never exceeding 24°C.

5.5 CONCLUSIONS

At sampling locality 9 the lower ASPT was ascribed to reduced habitat diversity. There was, however, an insufficient quantity and variety of water quality data to rule deterioration of water quality out as the negative influence on the ASPT. The influence of shortage of water quality data was encountered throughout this survey.

Similarities with the interpretation guidelines concerning some deterioration in water quality were found at sampling locality 26 (reduced water quality through rust deposit), and 35 (reduced water quality through silt deposits). It should be noted that silt and turbidity are considered as components of water quality.

Special cases were encountered at sampling locality 16 (multi channel system), and 36 (large pool). The evaluation of the results at these localities thus deviated from the interpretation guidelines.

Throughout the interpretation of the results it was apparent that habitat played a major role in the different scores obtained, as the results largely followed changes in habitat quality and availability. The interpretation of the SASS4 results is thus made considerably easier when this invertebrate biomonitoring method is accompanied with one or more habitat assessment methods, e.g. the Integrated Habitat Assessment System (IHAS) or the Habitat Quality Index (HQI). The IHAS was used to gain knowledge about the sampling habitats as well as stream conditions. It played a major role in determining the influence of the habitat on the SASS4 scores. It allows for quick identification of what biotopes were sampled. It also tries to link low SASS4 scores to the condition of the stream. It serves as a means of comparing the SASS4 scores for different localities by omitting aquatic habitat variability.

According to Chutter (1998), the ASPT is more important than the SASS4 sample score in interpreting the meaning of the results at high ranges of SASS4 scores. When interpreting the ASPT one considers both water quality and habitat diversity, as the value is calculated by dividing the SASS4 score with the total number of families identified. For example, consider a series of high SASS4 scores (as is the case in this study). When a lower ASPT is calculated, it would mean that most of the taxa that were sampled had low tolerance scores (thus a smaller SASS4 total to divide the number of families by). Such a shift in the community structure would be induced by deteriorating water quality, as SASS is based on the tolerance of aquatic benthic macroinvertebrates to reduced water quality. If the ASPT is higher, water quality has increased, as there is a higher occurrence of high scoring taxa. When chemical variables show a constant distribution and no physical disturbances are apparent, diminished ASPTs are mostly caused by an absence or reduction of a habitat where sensitive, high scoring taxa are usually found. This is the case at sampling locality 26 (Damani Dam pump). It should be remembered that the absence of certain habitats has a much larger impact on the SASS4 score measured than on the ASPT (Chutter, 1998). It remains evident that SASS scores, as well as ASPTs, are responsive to general environmental conditions and not only to changes in chemical water quality.

The importance of using ASPT when interpreting the scores obtained is also clearly demonstrated by Dallas (1995). It was found that the SASS4 score increased with sampling effort (i.e., the more samples that were taken at a locality, the higher the SASS4 score), whereas the ASPT changed very little with sampling effort.

Through monitoring the changes in the benthic invertebrate community with SASS4, and assessing the habitat conditions with IHAS, valuable knowledge can be gained about the type, seriousness, and temporal and spatial characteristics of a certain impact. It can thus be concluded that these techniques are important informative and regulatory tools in water resource management programs in South Africa. The contribution of SASS4 and IHAS to effective resource management and the assessment of the biotic integrity of a river are further investigated in chapter 6 and 7.

5.6 REFERENCES

CHUTTER, F. M. (1998) Research on the Rapid Biological Assessment of Water Quality Impacts in Streams and Rivers. WRC Report No. 422/1/98

DALLAS, H. F. (1995) *An Evaluation of SASS (South African Scoring System) as a Tool for the Rapid Bioassessment of Water Quality*. MSc Thesis. University of Cape Town, Cape Town

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Table 5A.1: Obtained SASS4 scores, number of families sampled, and ASPTs.

DATE	SAMPLING LOCALITY NO.	SAMPLE SCORE	NO. OF FAMILIES	ASPT
19/08/99	1	184	23	8.00
19/08/99	2	160	23	6.95
18/08/99	3	133	16	8.31
19/08/99	4	123	18	6.83
18/08/99	5	132	19	6.94
16/08/99	6	230	30	7.66
16/08/99	7	165	23	7.17
17/08/99	8	137	20	6.85
17/08/99	9	117	21	5.51
30/08/99	10	146	18	8.10
17/08/99	11	169	24	7.04
18/08/99	12	167	24	6.95
01/11/99	13	107	16	6.68
18/11/99	14	129	21	6.14
04/11/99	15	169	25	6.76
22/09/99	16a	179	26	6.88
22/09/99	16b	173	28	6.17
22/09/99	16c	101	20	5.00
22/09/99	16d	87	18	4.83
01/11/99	17	173	25	6.92
21/10/99	18	153	22	6.95
18/10/99	19	180	28	6.43
18/10/99	20	174	28	6.21
20/10/99	21	176	26	6.76
19/10/99	22	199	29	6.86
19/10/99	23	156	22	7.09
19/10/99	24	203	27	7.52
18/11/99	25	140	19	7.37
03/11/99	26	77	15	5.13
04/11/99	27	137	20	6.85
31/08/99	28	115	18	6.38
31/08/99	29	144	20	7.20
31/08/99	30	153	20	7.65
31/08/99	31	117	18	6.50
04/11/99	32	146	23	6.34
21/09/99	33	184	29	6.34
01/09/99	34a	140	18	7.78
01/09/99	34b	120	18	6.67
03/11/99	35	99	18	5.50
02/09/99	36a	209	25	6.80
02/09/99	36b	62	9	7.27
01/09/99	37	189	26	8.36
01/09/99	38	149	22	6.77
02/09/99	39	138	20	6.90
03/11/99	40	127	17	7.47
02/11/99	41	180	23	7.83
02/11/99	42	134	18	7.44

Table 5A.2a: Abundances of the different invertebrate families sampled.

Sampling locality no.	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
Hydra	0	0	0	1	0	0	0	A	0	0	0	0	0	0	0
Planaria	1	1	1	0	0	1	1	0	0	A	A	0	0	0	0
Oligochaeta	0	0	0	0	1	A	0	0	A	0	0	0	B	B	A
Hirudinea	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amphipoda	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Crabs	A	A	B	A	A	A	A	A	A	1	A	B	A	B	B
Shrimps	B	0	0	1	A	B	0	A	B	0	B	A	1	B	0
Hydrachnellae	B	0	B	A	B	A	B	B	0	B	B	A	0	0	A
Perlidae	A	A	0	0	0	A	0	0	0	0	1	A	A	1	A
Baetidae 1 sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Baetidae 2 spp	0	0	0	0	0	B	A	0	B	0	0	0	0	0	B
Baetidae >2 spp	B	A	B	B	B	0	0	B	0	B	B	B	B	B	0
Heptageniidae	A	A	A	0	0	0	0	0	B	A	A	B	A	0	A
Leptophlebiidae	0	A	B	0	B	B	B	A	0	B	B	A	0	0	1
Trichorythidae	A	0	B	B	A	B	B	0	0	A	0	0	0	0	0
Caenidae	A	0	0	A	B	B	B	A	A	A	B	1	0	0	0
Coenagriidae	B	0	A	A	0	1	1	A	B	0	A	A	A	A	A
Chlorocyphidae	A	A	A	A	1	A	1	0	A	A	A	A	0	A	0
Gomphidae	A	0	B	B	B	A	A	0	A	0	B	0	0	B	B
Aeshnidae	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Corduliidae	0	0	0	0	0	1	1	0	0	A	0	1	1	1	A
Libellulidae	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Notonectidae	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Pleidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Naucoridae	0	A	0	0	B	0	0	0	1	0	A	1	B	B	A
Nepidae	0	0	0	0	0	0	0	1	0	0	0	1	0	0	0
Corixidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gerridae	0	1	0	0	0	1	0	0	0	0	A	0	0	0	1
Veliidae	A	B	B	0	0	A	1	B	B	0	B	A	A	B	0
Hydropsychidae 1sp	0	0	0	0	0	0	0	0	1	0	0	0	A	0	0
Hydropsychidae 2spp	A	A	0	B	B	A	A	B	0	0	B	A	0	0	B
Hydropsychidae >2 spp	0	0	B	0	0	0	0	0	0	B	0	0	0	B	0
Philopotamidae	0	0	0	0	0	0	0	A	0	0	0	0	0	0	0
Cased caddis 1 type	1	0	0	0	A	0	0	0	0	0	0	0	0	0	0
Cased caddis 2 types	0	0	0	A	0	0	0	0	0	1	A	0	0	0	0
Cased caddis 3 types	0	B	A	0	0	0	0	B	B	0	0	0	0	0	B
Cased caddis 4 types	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cased caddis 5 types	0	0	0	0	0	A	0	0	0	0	0	0	0	0	0
Nymphulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae	0	0	0	0	0	0	0	0	0	0	0	0	0	1	A
Elmidae	0	0	0	0	0	0	A	0	0	0	0	0	1	A	B
Gyrinidae	B	0	B	A	0	A	B	1	B	A	A	0	0	B	A
Helodidae	0	0	A	0	0	1	A	0	0	0	0	0	0	0	0
Psephenidae	0	0	1	0	0	0	A	0	0	0	0	0	0	0	0
Tipulidae	A	A	B	0	A	B	B	B	A	1	B	A	1	0	A
Culicidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Simuliidae	B	B	B	B	A	B	B	B	A	B	B	A	A	B	B
Chironomidae	B	A	B	B	B	B	B	B	B	B	B	A	B	B	B
Ceratopogonidae	1	0	1	1	1	1	0	A	0	0	A	A	0	A	A
Tabanidae	1	0	1	A	A	0	0	0	0	A	0	B	0	B	A

Table 5A.2a: (continued)

Syrphidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athericidae	A	A	B	A	B	B	B	A	0	A	A	B	A	0	1
Lymnaeidae	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Melaniidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbidae	0	0	0	0	0	1	0	0	1	0	0	A	0	A	0
Physidae	0	0	0	0	0	0	0	0	1	0	0	1	0	1	0
Ancylidae	0	0	A	0	0	1	1	0	0	0	0	0	0	0	0
Sphaeridae	0	0	0	0	0	0	0	0	0	0	A	0	0	0	1

Table 5A.2b: Abundances of the different invertebrate families sampled.

Sampling locality no.	16a	b	c	d	17	18	19	20	21	22	23	24	25	26	27	28
Hydra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planaria	0	0	0	0	0	0	1	A	A	0	0	1	1	0	0	0
Oligochaeta	B	1	A	B	1	0	B	A	B	1	A	0	0	A	0	A
Hirudinea	0	0	0	A	0	0	0	0	1	0	0	0	0	0	0	0
Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crabs	A	A	0	1	A	B	1	0	0	0	0	0	A	A	B	1
Shrimps	A	A	0	0	A	A	1	A	B	B	A	B	0	0	0	0
Hydrachnellae	A	0	0	A	0	A	0	1	0	0	0	0	A	0	B	B
Perlidae	B	A	0	0	A	A	A	A	A	A	B	A	0	0	0	0
Baetidae 1 sp	0	0	A	B	0	0	0	0	0	0	0	0	0	0	0	0
Baetidae 2 spp	0	0	0	0	0	0	0	B	0	B	0	B	0	A	0	A
Baetidae >2 spp	B	B	0	0	B	B	B	0	A	0	B	0	B	0	A	0
Heptageniidae	B	B	1	A	B	A	A	A	B	A	B	A	0	0	A	1
Leptophlebiidae	A	A	0	0	A	A	A	0	A	A	A	A	0	0	0	0
Trichorythidae	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	A
Caenidae	0	A	0	1	A	A	0	A	A	B	1	B	0	0	0	0
Coenagriidae	A	A	A	1	A	0	A	0	A	A	A	A	0	A	0	A
Chlorocyphidae	A	0	0	0	A	1	1	0	A	A	0	0	B	0	B	A
Gomphidae	A	B	1	0	A	A	A	B	A	B	B	A	B	0	A	B
Aeshnidae	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0
Corduliidae	A	A	1	B	0	1	B	A	0	A	1	A	0	A	A	B
Libellulidae	0	0	0	0	0	0	1	0	0	0	0	0	1	0	0	0
Notonectidae	0	0	1	B	0	0	0	A	0	A	0	1	1	0	0	0
Pleidae	A	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Naucoridae	A	B	0	1	B	B	A	B	B	A	B	A	0	A	A	0
Nepidae	0	0	A	0	0	0	0	0	0	A	0	0	0	1	0	0
Corixidae	1	1	A	B	0	0	A	A	0	0	0	0	0	0	0	0
Gerridae	0	0	1	A	0	0	0	1	0	A	A	A	A	0	0	0
Veliidae	B	B	A	A	B	A	B	1	A	A	B	B	B	B	B	B
Hydropsychidae 1sp	0	B	0	0	0	A	A	0	1	A	0	0	0	A	0	0
Hydropsychidae 2spp	A	0	0	0	0	0	0	A	0	0	A	A	0	0	B	0
Hydropsychidae >2 spp	0	0	0	0	B	0	0	0	0	0	0	0	B	0	0	A
Philopotamidae	0	0	0	0	0	0	0	0	B	A	A	A	1	0	0	0
Cased caddis 1 type	0	1	0	0	0	A	A	0	0	0	0	0	0	0	0	1
Cased caddis 2 types	A	0	A	0	B	0	0	0	A	0	A	0	0	A	A	0
Cased caddis 3 types	0	0	0	0	0	0	0	A	0	A	0	0	0	0	0	0
Cased caddis 4 types	0	0	0	0	0	0	0	0	0	0	0	0	B	0	0	0
Cased caddis 5 types	0	0	0	0	0	0	0	0	0	0	0	A	0	0	0	0
Nymphulidae	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Dytiscidae	0	0	1	A	A	0	0	A	1	A	0	A	0	0	0	0
Elmidae	A	1	0	0	1	A	A	A	A	A	B	B	0	0	1	0
Gyrinidae	0	B	0	0	B	A	B	A	A	0	0	0	B	A	0	0
Helodidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Psephenidae	0	0	0	0	0	0	0	0	0	0	0	0	A	0	0	0
Tipulidae	A	A	0	1	A	0	1	0	0	0	0	0	1	1	B	1
Culicidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	A	0
Simuliidae	B	B	0	A	A	A	B	A	B	A	B	B	A	0	B	B
Chironomidae	B	B	A	B	A	A	B	B	B	B	A	A	A	B	B	B
Ceratopogonidae	0	A	B	0	B	0	A	0	0	0	1	A	0	A	A	A
Tabanidae	A	A	A	0	A	0	0	A	A	A	A	A	0	0	A	A

Table 5A.2b: (continued)

Syrphidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Athericidae	A	A	0	0	0	1	0	1	0	A	0	1	0	0	B	A
Lymnaeidae	1	A	1	0	0	0	0	A	0	0	0	0	0	0	0	0
Melaniidae	0	0	1	0	0	0	1	B	0	0	0	0	0	0	0	0
Planorbidae	0	1	0	0	0	0	0	A	1	B	0	A	0	0	B	0
Physidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ancylidae	0	A	0	0	A	1	1	0	0	1	0	0	0	0	0	0
Sphaeridae	A	A	1	0	A	B	1	B	B	A	1	A	0	0	0	0

Table 5A.2c: Abundances of the different invertebrate families sampled.

Sampling locality no.	29	30	31	32	33	34a	b	35	36a	b	37	38	39	40	41	42
Hydra	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planaria	1	0	A	0	0	0	0	0	A	0	0	0	0	0	0	0
Oligochaeta	0	0	A	1	A	0	0	0	1	1	1	0	A	0	1	0
Hirudinea	0	A	0	0	0	0	A	0	0	0	0	0	0	0	0	0
Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crabs	1	A	A	B	A	0	0	A	1	0	A	1	1	1	0	A
Shrimps	0	0	1	0	A	0	0	0	0	0	0	0	1	0	0	0
Hydrachnellae	B	0	0	A	A	0	0	0	A	0	B	B	B	B	0	B
Perlidae	0	B	0	A	A	A	0	0	1	0	A	0	A	0	A	B
Baetidae 1 sp	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Baetidae 2 spp	0	0	0	0	0	B	B	A	0	A	0	0	A	B	B	0
Baetidae >2 spp	B	B	B	B	B	0	0	0	B	0	B	B	0	0	0	B
Heptageniidae	0	B	B	1	A	0	0	0	A	0	B	0	1	1	B	A
Leptophlebiidae	A	A	1	0	0	A	0	0	0	0	0	0	0	0	1	A
Trichorythidae	B	B	0	0	0	0	A	0	A	0	A	A	0	0	0	0
Caenidae	A	B	0	A	A	A	0	0	1	0	A	A	A	0	1	0
Coenagriidae	0	A	0	A	A	A	A	1	A	1	0	A	1	A	A	A
Chlorocyphidae	1	A	0	A	B	A	0	A	A	0	A	B	A	A	1	0
Gomphidae	0	0	0	B	1	A	A	A	B	0	B	B	0	A	A	B
Aeshnidae	0	0	0	0	0	0	0	1	0	0	1	A	1	0	0	0
Corduliidae	0	0	0	0	A	A	A	A	B	0	B	B	0	A	1	1
Libellulidae	1	B	A	0	0	1	A	0	0	0	0	0	0	0	0	0
Notonectidae	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Pleidae	0	0	0	0	0	0	0	0	0	A	0	0	0	0	A	0
Naucoridae	0	0	1	B	B	0	1	0	1	0	A	A	A	A	0	A
Nepidae	0	0	0	0	0	0	0	1	1	0	A	0	0	0	0	0
Corixidae	0	0	0	0	A	0	0	0	0	0	0	A	0	0	0	0
Gerridae	0	0	0	0	A	0	A	0	0	A	A	0	0	0	A	0
Veliidae	1	1	A	B	B	0	A	A	B	A	A	A	A	A	A	B
Hydropsychidae 1sp	0	0	0	0	0	A	0	0	0	0	0	0	1	0	0	0
Hydropsychidae 2spp	0	B	A	B	A	0	0	B	A	0	A	A	0	0	A	B
Hydropsychidae >2 spp	B	0	0	0	0	0	0	0	0	0	0	0	0	A	0	0
Philopotamidae	0	0	0	0	0	0	A	0	0	0	0	0	0	0	0	0
Cased caddis 1 type	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Cased caddis 2 types	B	0	0	0	A	A	A	A	0	0	B	A	0	1	0	A
Cased caddis 3 types	0	B	0	0	0	0	0	0	0	0	0	0	A	0	0	0
Cased caddis 4 types	0	0	0	0	0	0	0	0	0	A	0	0	0	0	A	0
Cased caddis 5 types	0	0	0	0	0	0	0	0	B	0	0	0	0	0	0	0
Nymphulidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae	0	0	0	0	0	A	0	0	0	1	0	0	0	0	0	1
Elmidae	0	0	0	A	0	0	0	0	A	0	0	0	0	A	1	B
Gyrinidae	B	A	A	A	B	0	A	1	A	0	B	B	0	0	0	0
Helodidae	0	0	0	0	0	1	1	0	1	0	1	0	0	0	0	0
Psephenidae	0	0	0	0	0	A	0	0	0	0	0	0	0	0	0	0
Tipulidae	1	0	0	A	A	A	0	1	1	0	0	0	0	0	1	0
Culicidae	0	0	0	A	0	0	0	0	0	0	0	0	0	0	0	0
Simuliidae	B	B	B	B	B	B	1	B	B	0	A	B	0	B	A	B
Chironomidae	B	B	B	B	B	B	C	B	B	A	B	B	B	B	B	B
Ceratopogonidae	1	1	1	0	A	A	0	0	A	0	1	B	B	1	0	0
Tabanidae	1	0	A	0	A	0	0	1	0	0	0	0	0	0	1	A

Table 5A.2c: (continued)

Syrphidae	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Athericidae	A	A	A	A	A	A	A	0	A	0	B	A	A	1	A	0
Lymnaeidae	0	0	0	0	0	0	0	1	0	0	0	0	1	0	0	0
Melaniidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Planorbidae	0	0	0	0	A	0	0	0	0	0	1	1	0	0	0	0
Physidae	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Ancylidae	0	1	0	1	A	0	0	0	0	0	0	A	0	0	1	0
Sphaeridae	0	0	0	1	A	0	0	0	0	0	0	0	0	0	0	0

Table 5A.3: Obtained IHAS scores.

Sampling locality no.	Sampling Habitat	Stream Condition	Total IHAS Score	Adjustment score
1	37	41	78	18
2	30	36	66	25
3	31	27	58	24
4	45	35	80	20
5	30	30	60	25
6	34	35	69	21
7	26	29	54	29
8	30	34	64	25
9	25	34	59	30
10	34	31	65	21
11	41	35	76	14
12	44	33	77	11
13	41	30	71	14
14	37	31	68	18
15	47	33	80	8
16a	46	29	75	9
16b	48	33	81	7
16c	18	20	38	37
16d	17	24	41	38
17	48	26	74	7
18	41	30	71	14
19	37	43	80	18
20	31	39	70	24
21	41	33	74	14
22	34	37	71	21
23	36	31	67	19
24	36	29	65	19
25	35	44	79	20
26	41	35	76	14
27	41	35	76	14
28	28	38	66	27
29	39	31	70	16
30	40	34	74	15
31	41	27	68	14
32	37	28	65	18
33	37	31	68	18
34*	36	39	75	19
35	44	32	76	11
36*	50	35	85	5
37	46	41	87	9
38	41	29	70	14
39	44	24	68	11
40	39	33	72	16
41	41	30	71	14
42	45	25	70	10

*IHAS was only performed once at sampling locality 34 and 36.

Table 5A.4: Adjusted SASS4 scores

Sampling locality no.	SASS SCORE	Adjustment score	Adjusted SASS Score
1	184	18	202
2	160	25	185
3	133	24	157
4	123	20	143
5	132	25	157
6	230	21	251
7	165	29	194
8	137	25	162
9	117	30	147
10	146	21	167
11	169	14	183
12	167	11	178
13	107	14	121
14	129	18	147
15	169	8	177
16a	179	9	188
16b	173	7	180
16c	101	37	138
16d	87	38	125
17	173	7	180
18	153	14	167
19	180	18	198
20	174	24	198
21	176	14	190
22	199	21	220
23	156	19	175
24	203	19	222
25	140	20	160
26	77	14	91
27	137	14	151
28	115	27	142
29	144	16	160
30	153	15	168
31	117	14	131
32	146	18	164
33	184	18	202
34a	140	19	159
34b	120	19	139
35	99	11	110
36a	209	5	214
36b	62	5	67
37	189	9	198
38	149	14	163
39	138	11	149
40	127	16	143
41	180	14	194
42	134	10	144

Table 5A.5: Measured chemo-physical variables.

SAMPLING LOCALITY NO.	TEMPERATURE (C)	pH	CONDUCTIVITY (µs/m)
1	15.0	8.0	40
2	15.0	8.0	40
3	NM	8.0	100
4	15.5	8.3	60
5	16.0	7.7	70
6	NM	8.3	80
7	NM	8.1	120
8	NM	8.0	120
9	NM	8.1	100
10	16.0	8.0	70
11	16.0	8.1	90
12	17.0	8.0	100
13	24.0	7.7	150
14	NM	NM	NM
15	NM	NM	NM
16a	19.0	8.0	120
16b	19.0	8.0	120
16c	19.0	8.0	120
16d	19.0	8.0	120
17	24.0	7.8	130
18	24.0	8.0	110
19	23.0	7.8	110
20	24.0	8.2	110
21	24.0	8.2	130
22	24.0	8.5	110
23	24.0	8.3	110
24	24.0	8.0	140
25	NM	NM	NM
26	23.0	NM	NM
27	NM	NM	NM
28	18.0	7.3	30
29	18.0	7.6	30
30	NM	NM	NM
31	18.5	7.8	90
32	NM	NM	NM
33	NM	7,8	90
34a	NM	NM	NM
34b	NM	7.9	20
35	NM	NM	NM
36a	NM	NM	NM
36b	NM	NM	NM
37	NM	NM	NM
38	NM	NM	NM
39	NM	NM	NM
40	NM	NM	NM
41	NM	NM	NM
42	NM	NM	NM

NM: Not Measured

CHAPTER 6

IMPLEMENTATION OF RESOURCE DIRECTED MEASURES

6.1 INTRODUCTION

The Resource Directed Measures (RDM) procedure for the protection of water resources provides the course of action for the preliminary determinations of the class, Reserve and Resource Quality Objectives (RQO) for water resources, as specified in sections 14 and 17 of the South African National Water Act (Act 36 of 1998).

In order to ensure sustainable utilisation, this policy argues that the resilience of a water resource should be maintained above a certain base level of ecological integrity and function, which is termed the Resource Base. Brewer (1994) defines resilience as the tendency of an ecosystem to return to its original state after a perturbation.

The Reserve is defined in terms of the quality and quantity of water which are needed to protect basic human needs, and the structure and function of ecosystems so as to secure ecologically sustainable development and utilisation (DWAF, 1999).

The requirements of the Reserve for a particular water resource are described as Resource Quality Objectives, which are a rigorous numeric or descriptive statement, given in measurable, enforceable terms. According to DWAF (1999), the Objectives have four critical components, to cover each of the aspects of ecological integrity, which are necessary for protection of the Resource Base:

- Requirements for water quantity, stated as Instream Flow Requirements (IFR) for a river reach or estuary, or water level requirements for standing water or groundwater. These are determined according to current procedures for assessing IFR, namely the Building Block Methodology (BBM);

- Requirements for water quality, which are determined on the basis of current guidelines and procedures as set out in the South African Water Quality Guidelines;
- Requirements for habitat integrity, which encompass the physical structure of instream and riparian habitats, as well as the vegetation aspects; and
- Objectives for biotic integrity, which reflect the health, community structure and distribution of aquatic biota.

The process of determining the Resource Quality Objectives is preceded and facilitated by the implementation of a protection-based classification system, where water resources are grouped into classes representing different levels of protection. Refer to Table 6.1-6.3. According to DWAF (1999):

- Table 6.1 shows an example of how classification of a water resource might be used to set water environment objectives which reflect an agreed balance between protection and utilisation.
- Table 6.3 describes the characteristics of the ecological integrity protection classes A through D, and resource quality objectives can be derived in order to protect and maintain those characteristics.
- In a similar manner, the fitness for use classes shown in Table 6.2 can be used as a basis for deriving numerical objectives to maintain certain water quality characteristics.

Table 6.4 describes the four levels of RDM determinations and the flow management plan, as well as the rules for selection of the appropriate level, which depend on the sensitivity of a water resource, the scale and degree of the impact of proposed water uses, and the urgency for a Reserve determination.

This chapter includes a brief summary of the Resource Directed Measures process. The reader is referred to the documents published by DWAF in September 1999, namely Water Resource Protection Policy Implementation and the relevant appendices, for further details concerning the methodology. The evaluation of a facet of a desktop RDM study which was performed for the Luvuvhu River, namely the Present Ecological Status, follows in the next chapter.

Table 6.1: A classification approach to balancing the requirements of protection and utilisation (DWAF, 1999).

Water resource	Ecosystem protection class	Desired status for domestic use	Desired status for agricultural (irrigation) use	Desired status for recreational use	Classification
River X, reach 1	A	Class I	Class II	full contact	$Ad_{I}a_{II}r_{f}$
River X, reach 2	B	Class II	Class III	intermediate contact	$Bd_{II}a_{III}r_{I}$
River X, reach 3	B	Class II	Class IV	intermediate contact	$Bd_{II}a_{IV}r_{I}$

Thus River X, reach 2 would have a classification of $Bd_{II}a_{III}r_{i}$. This means that

B: the ecological integrity status of that reach would be maintained at class B (see Table 6.3)

d_{II}: the water quality would be fit for domestic use with conventional treatment (see Table 6.2);

a_{III}: the water quality would be fit for irrigation of moderately tolerant crops, depending on site-specific soil conditions;

r_{I}: the water resource would be fit for intermediate contact recreation.

Table 6.2: Water quality "fitness for use" classes currently used in South Africa (DWAF, 1999).

Water use	Categorisation	Description
Domestic	Class 0	Water of ideal quality, which has no health or aesthetic effects and which is suitable for lifetime use without negative effects. No treatment necessary.
	Class I	Water of good quality, suitable for lifetime use with few health effects. Aesthetic effects may be apparent. Home treatment will usually be sufficient.
	Class II	Water which poses a definite risk of health effects, following long term or lifetime use. However, following short-term or emergency use, health effects are uncommon and unusual. Treatment will be required in order to render the water fit for continued use.
	Class III	Water is unsuitable for use , especially by children and the elderly, as health effects are common. Conventional or advanced treatment necessary
Recreation	Full contact	Water is suitable for recreation which involves full body immersion and the likelihood of ingestion of water.
	Intermediate contact	Water is suitable for activities such as water-skiing, canoeing, sailing, and those which involve paddling and wading, with only occasional immersion.
	Non-contact or aesthetic	Water unsuitable for direct contact, but meets criteria for scenic and aesthetic appreciation.
Irrigation	Class I	No yield reduction for even the most sensitive crops. Safe for surface application or foliar wetting systems. Leaching fraction ≤ 0.10 No impact on soils.
	Class II	Yield of at least 95% for moderately sensitive crops. Safe for surface application. Leaching fraction ≤ 0.10 No impact on moderately sensitive soils.
	Class III	Yield of at least 90% for moderately tolerant crops. Safe for surface application. Leaching fraction ≤ 0.15 Management practices required for sensitive soils.
	Class IV	Yield of at least 80% for moderately tolerant crops. No foliar wetting systems. Leaching fraction ≤ 0.20 Intensive management practices required for sensitive soils – not economical.
Stock watering	Target guideline range	No adverse effects on stock
	Outside target guideline range	Adverse effects, depending on the specific constituent, the type of livestock, previous adaptation.
Aquaculture	Target guideline range	No adverse effects on aquaculture species
	Outside target guideline range	Adverse effects, depending on the specific constituent, the species, previous adaptation.

Table 6.3: Proposed framework for setting ecological resource quality objectives on the basis of a classification system (DWAF, 1999).

Class	Water quantity	Water quality	Instream habitat	Riparian habitat	Biota
A	Natural variability and disturbance regime: Allow negligible modification.	Negligible modification from natural. Allow negligible risk to sensitive species. Within Aquatic Ecosystems TWQR for all constituents.	Allow negligible modification from natural conditions. Depends on the instream 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 negligible modification (eg 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 instream 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 instream 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 instream 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 instream 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 instream 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 instream 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.

Table 6.4: Levels of RDM procedures for various levels of RDM determinations (DWAF, 1999).

Level	**Term**	**Characteristics**	**Use**
1	*Desktop estimate*	Very low confidence, about 2 hours per water resource	For use in National Water Balance Model only
2	*Rapid determination*	Low confidence; desktop + quick field assessment of present status, takes about 2 days	Individual licensing for small impacts in unstressed catchments of low importance & sensitivity; compulsory licensing "holding action"
3	*Intermediate determination*	Medium confidence, specialist field studies, takes about 2 months	Individual licensing in relatively unstressed catchments
4	*Comprehensive* determination	Relatively high confidence, extensive field data collection by specialists, takes 8-12 months	All compulsory licensing. In individual licensing, for large impacts in any catchment. Small or large impacts in very important and/or sensitive catchments.
	Flow management plan	Acknowledges present operating constraints (mostly structural) on a river; modified operating rules are drawn up between the management agency and RDM study team, which will result in a more environmentally friendly flow regime, as far as possible.	For use in highly regulated systems where present flow control structures do not have outlets from which releases can be made to provide for the water quantity component of the Reserve.

6.2 RDM METHODOLOGY

The RDM determination is a seven-step generic methodology, within which all four levels of RDM procedures (desktop, rapid, intermediate and comprehensive) fit.

Step 1: Initiate the RDM study

Identify significant water resources

The term "significant" relates to the geographic extent of the water resource for which a class, Reserve and RQOs must be determined (DWAF, 1999).

Delineate geographical boundaries for the RDM study

It is practical to delineate water resources or parts of water resources for water use planning, allocation and licensing purposes. It should also be at a scale which allows effective everyday management of the water resource itself (DWAF, 1999).

Select appropriate level of RDM determination

Refer to Table 6.4

Establish study team composition

The composition of the study team depends on the level of the RDM study which is required, and the types of water resources which are included within the study boundaries

Step 2a: Determine the ecoregional type of each resource unit

Ecologically homogeneous resource units

Once the ecologically homogenous resource units are determined, this knowledge is then used in steps 3, 4, 5 and 6 to guide expert judgement regarding what the appropriate numerical water quantity and quality requirements might be for achieving different levels of protection of that particular water resource.

Factors determining ecoregional type

According to DWAF (1999), ecoregional type is determined by

- Major physiographic factors, such as altitude, latitude, aspect, slope, climate zone;

- Geological factors, which determine geochemical signature as well as aspects of geomorphology and habitat;
- Regional natural hydrological characteristics, such as annual precipitation, seasonality and variability of flow;
- Major natural vegetation types (such as bushveld, lowveld etc);
- Biotic factors, including the natural occurrence of certain kinds of organisms such as fish, invertebrates, plants and algae.

Ecoregional mapping of South Africa

The DWA&F has recently initiated a project to map and delineate ecoregions in South Africa.

Step 2b: Delineate resource units within the study area

Breaking down the catchment into resource units

Resource units are significantly different from one-another, and will thus have their own specification of the Reserve, and the geographic boundary of each should be delineated.

Ecoregions

The breakdown of a catchment into water resource units for the purpose of determining the Reserve for rivers is done primarily on a biophysical basis, according to the occurrence of different ecological regions (ecoregions) within the catchment (DWAF, 1999).

Geohydrological response units

For groundwater, water resource units are initially defined on the basis of geohydrological response units.

Resource units related to land use and management needs

A further breakdown of an ecoregion could be done on the basis of major land uses.

Delineation of resources in the Reserve determination notice

The Government Gazette notice of the Reserve determination for a particular resource would refer to a map with the relevant water resource unit boundaries marked, and may be supported by map references (for example "the reach of the Crocodile River from the point A,B on the farm <name> to the point X,Y on the farm <name>") (DWAF, 1999). If a notice was published containing the Reserve determination for an entire catchment, the notice would contain a list of delineated resource units within the catchment, together with the class, Reserve and RQOs applicable to each resource unit.

Step 2c: Select survey sites within the study area

Site selection for specialist surveys

The selection of sites for specialist surveys depends on ecoregional types, representivity and suitability for hydraulic calibration and accessibility.

According to DWAF (1999), this is one of the most important steps in the RDM determination, as the confidence in the determination of the Reserve, especially, is very much dependent on the selection of suitable study sites.

Site selection related to resource units

For water quantity and water quality Reserve determination, it may be adequate to select sites, which cover more than one ecoregion. For determination of resource quality objectives for habitat and biota, the RQOs may need to be determined separately for each ecoregion.

Step 3: Determine the reference conditions for each resource unit

What are reference conditions?

Reference conditions describe the natural unimpacted characteristics of a water resource, and further quantitatively describe the ecoregional type, specific to a particular water resource (DWAF, 1999).

Defining reference conditions for surface water ecosystems

According to DWAF (1999), reference conditions for surface water ecosystems involve:

- Water quantity - the amount, timing, pattern and levels of flow, including seasonal and inter-annual variability, flood and drought cycles;
- Water quality - the concentrations of key water quality constituents, including their seasonal and inter-annual variability, and going as far as diurnal patterns of variability for constituents such as temperature, dissolved oxygen and pH where relevant;
- the geomorphologic characteristics of instream and riparian habitat, as well as the vegetation aspects of habitat;
- the character, composition and distribution of aquatic biota.

Defining reference conditions for groundwater resources

The procedure for derivation of reference conditions for groundwater resources requires additional work, and will be addressed in the future

Natural unimpacted conditions - a stable baseline

The reference conditions are set on the basis of natural unimpacted conditions, as this serves as the most stable baseline available to performing steps 4, 5, and 6 of the RDM determination.

Using expert judgement to determine reference conditions

When unimpacted sites are unavailable, expert judgement, local knowledge, historical data and analysis of measured historical trends will be necessary to build up a "picture" of the probable reference conditions, within broad confidence limits.

"Resetting" reference conditions

Situations may occur where the water resource has been modified to such an extent, and in such a manner, that ecosystem structure, functions and processes have been irreversibly changed, but where the ecosystem is still healthy and requires protection. This would justify the resetting of the reference conditions so as to more accurately reflect the new ecological characteristics.

Step 4a: Assess the present status of the resource units

Assessment of resource status and resource use

Step 4 entails a full present status assessment of resource quality (water quantity, water quality, habitat and biota) in terms of the degree of modification from reference conditions, as well as an assessment of current and projected water uses, land uses and socio-economic conditions.

Ecological assessment categories

There are 6 ecological present status categories for rivers, namely A through F. Each represents a broad band of "degree of modification" from reference conditions. Category A thus indicates negligible modification, closely approximating natural conditions, and category F indicates critical modification.

Purpose of the present status assessment

According to DWAF (1999), the present status assessment is required for two purposes:

- Firstly to assess the degree of modification, (and hence the current degree of risk of irreversible damage), and if possible to identify whether resource quality is stable within a particular assessment category, or whether the resource is currently degrading due to past or present impacts; and
- Secondly to identify what may be achievable in terms of the future management class, in order to rule out unrealistic options when setting the management class in step 5. Sometimes structural modifications to the resource (such as dams or urban development), or short-term needs for economic development may be such that a higher class than the present one can not be practically achieved in the short to medium term.

Step 4b: Assess the importance and sensitivity of the resource units

Ecological importance and sensitivity

Ecological importance of a river is an expression of its importance to the maintenance of ecological diversity and functioning on local and wider spatial scales (DWAF, 1999). Ecological sensitivity (or fragility) refers to the system's resilience (i.e. the ability of the system to recover from a disturbance - Roux *et al*, 1997).

In determination of RDM, the following ecological aspects are considered as the basis for the estimation of ecological importance and sensitivity (DWAF, 1999):

- The presence of rare and endangered species, unique species (i.e. endemic or isolated populations) and communities, intolerant species and species diversity
- Habitat diversity, including specific habitat types such as reaches with a high diversity of habitat types, i.e. pools, riffles, runs, rapids, waterfalls, riparian forests, etc.
- The importance of the particular resource unit (e.g. river or reach of river) in providing connectivity between different sections of the whole water resource, i.e. whether it provides a migration route or corridor for species.
- The presence of conservation areas or relatively natural areas along the river section
- The sensitivity (or fragility) of the system and its resilience (i.e. the ability to recover following disturbance) of the system to environmental changes is also considered. Consideration of both the biotic and abiotic components is included here.

Social importance

Some of the aspects to be included in the assessment of socio/cultural importance are the extent to which people are dependent on the natural ecological functions of the water resource for water for basic human needs (sole source of supply), and recreational, historical, archaeological, intrinsic and aesthetic value of the water resource.

Economic importance

The economic value of a water resource is traditionally assessed in terms of the amount of water which can be abstracted for offstream use (DWAF, 1999). Typical indicators include the number and value of jobs generated by the use of the water, or the amount of revenue generated.

Why we need to know about importance and sensitivity

The importance and sensitivity of a water resource is used to guide or influence the decision on the level of protection required, which in turn determines the management class which should be assigned.

Step 5: Set the management class for each resource unit

Setting a management goal

Step 5 entails the selection of an appropriate management class as the target for long term protection and management of the resource unit. This could be the same as the present status category, or it could be set higher if improvement of resource quality is required.

Testing the implications of selecting a particular management class

The implications for the Reserve of selecting a particular management class can be established and various scenarios tested before the final class is set.

Classification rules for ecosystems, basic human needs and water users

According to DWAF (1999), the water resource classification system, when fully developed, will have three aspects: it will set out clear rules for

- protection of basic human needs
- protection of aquatic ecosystems
- water user requirements

When the full classification system is implemented, then the classification step in the Reserve determination process will require integration of the requirements of basic human needs, ecosystems and water users.

Acceptable ecosystem management classes

Of the 6 ecological status categories A to F, the first four (A through D) are matched to the ecological management classes in the water resource classification system, as only these classes are acceptable ecological management classes. Classes E and F indicate an unacceptable high risk of irreversible degradation.

Criteria for selecting a future management class

According to DWAF (1999), criteria for selecting a class for a water resource include:

- the sensitivity of the resource to impacts of water use (whether due to ecological sensitivity, or the sensitivity of downstream water users);
- the importance of the resource, in ecological, social/cultural or economic terms;
- what can be achieved towards improvement of resource quality, given that some prior impacts or modifications may not be practically reversible due to technical, social or economic constraints.

A consultative process for classifying a resource

For intermediate and comprehensive RDM determinations, the actual process of assigning a management class to a specific water resource will be a consultative one.

This consultative process must address long term protection requirements as well as accounting for economic and social issues, in reaching a balance between protection for long term sustainability on one hand, and short to medium term development needs on the other.

Different levels of decision-making process

According to DWAF (1999), the following rules for selection of the management class apply for the different RDM determination levels:

- **Setting of the ecological management class in desktop estimates:**

The management class is directly translated from the ecological importance and sensitivity rating.

- **Setting of the ecological management class in rapid determinations:**

The management class is translated directly from the ecological importance and sensitivity rating, but moderated by a shortened intermediate habitat integrity assessment (not the complete intermediate present status assessment).

- **Setting of the ecological management class in intermediate determinations:**

According to DWAF (1999), the so-called "default rule" applies, where after an intermediate present status assessment, the management class is set in relation to the present status, but at a level which represents a goal of no further degradation for resources which are slightly to largely modified, and at least a move toward improvement for resources which are critically modified:

Present Status Assessment Category	Ecological management class assigned
A	A
B	B
C	C
D	C
E or F	D

- **Setting of the management class in comprehensive determinations**

A formal process of consultation and participation should lead up to the decision on which management class will be set for the resource. The guidelines for consultation are included in the Classification System manual.

Step 6a: Quantify the Reserve for each resource unit

Direct link from the management class

Step 6 involves the quantification of the Reserve (water quantity and water quality) for a particular water resource, according to the rules associated with the management class which has been assigned to the resource unit in step 5.

Rules for deriving site-specific numerical values

The classification system rules for setting the Reserve and resource quality objectives will not be a "set of numbers" valid for all water resources: in most cases, the rules are rigorous procedures for deriving site-specific numerical objectives which are appropriate for the reference conditions of a particular resource (DWAF, 1999).

Integration of the water quantity and water quality components of the Reserve

Determination of the water quantity and water quality components of the Reserve must be carried out in an integrated manner.

Integration and matching for all the resource units in the study area

The RDM requirements of each resource unit must be matched with those for the adjacent resource units.

Step 6b: Set resource quality objectives for each resource unit

RQOs for instream and riparian habitat

Numerical or narrative RQOs for instream habitat and riparian habitat should be set which would ensure the appropriate extent, distribution, type and integrity of these habitats.

RQOs for aquatic biota

RQOs for aquatic biota can include measures of biotic integrity such as the SASS (invertebrate) and fish integrity scores. RQOs in terms of scores can be set for biotic integrity but the achievement of these objectives can only be assured through maintenance of an appropriate abiotic template (water quantity, water quality and habitat integrity) (DWAF, 1999).

RQOs for instream or land-based activities

The NWA also provides for the option of setting RQOs, which can be narrative or numerical, to ensure that instream or land-based activities, which may affect the water quantity or quality of a water resource, are regulated or prohibited (DWAF, 1999).

Step 7: Design an appropriate resource monitoring programme

Objectives of post-RDM monitoring

According to DWAF (1999), these objectives entail the following:

- To collect data to improve the confidence of a future RDM determination at the next level (e.g. to prepare for a future comprehensive determination if the present determination was at intermediate level);
- To monitor the response of the aquatic ecosystem to the Reserve and RQO that were set, to check that the Reserve and RQO do actually provide the level of protection required by the selected management class;
- To monitor resource quality status in order to ascertain whether management actions are adequate to achieve compliance with the requirements of the Reserve and RQO.

6.3 REFERENCES

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CHAPTER 7

EVALUATION OF SASS4 IN THE DETERMINATION OF THE DESKTOP RDM FOR THE LUVUVHU RIVER

7.1 INTRODUCTION

As stated in Table 6.4, the desktop determination of RDM was developed to serve the National Water Balance Model (NWBM), and thus only provides an estimate of the water quantity component of the ecological Reserve. Flow requirements are based on an index of the ecological importance and sensitivity (EISC) of rivers in quaternary catchments (delineated boundaries) (Kleynhans *et al*, 1998). This index is then used as an indicator of the default ecological management class (DEMC) that can then be compared to the present ecological status category (PESC). According to DWAF (1999), the desktop PESC assesses attributes that modify instream and riparian habitat and impact on native biota (refer to Table 7.1). Each of these attributes is then scored according to scoring and rating guidelines and the mean is then calculated. This is used to place the main river in the quaternary catchment in a particular present ecological status class. The DEMC relates to a default ecological status class (DESC), which in theory, should be assigned to a resource, given an indicated level of ecological importance and sensitivity (DWAF, 1999).

Further, an attainable EMC can be derived by assessing the deviation of the PESC from the DESC in terms of the practicality of restoring a system, following an assessment of the changes that have occurred. The total procedure followed to determine the AEMC is indicated in Figure 7.1.

Table 7.1: Attributes to be assessed for determination of present ecological status (DWAF, 1999).

Assessment Attributes	Considerations for Assessment of Attributes
Flow	Relative deviation from the expected natural - modification/ deterioration of habitat due to abstraction and/or flow regulation.
Inundation	Relative degree of inundation by weirs (or similar structures) and impoundments - loss of instream habitat and the riparian zone as well as the possible fragmentation effect on biological populations and communities.
Water quality	Water quality (relative degree of modification from the expected natural, and its perceived biological impact and significance).
Stream bed condition	Stream bed condition (relative degree of modification as caused by disturbances such as sedimentation, covering by excessive algal growth related to eutrophication, etc.).
Introduced instream biota	Species involved, their characteristics and severity of impact on native biota - i.e., impact on physical habitat and competition, predation.
Riparian or stream bank condition	Riparian or stream bank condition (relative deviation from the expected natural situation as indicated by disturbances such as removal of vegetation, invasive vegetation, erosion, etc.).

EISC:	**DEMC:**	**DESC:**
Very high	→ No human induced hazards	→Class A: Unmodified natural
High	→ Small risk allowed	→Class B: Largely natural
Moderate	→ Moderate risk allowed	→Class C: Moderately modified
Low/marginal	→ Large risk allowed	→Class D: Largely modified

PESC:

PESC: POSSIBLE ATTAINABLE IMPROVEMENT:

Acceptable range of AEMC:

Category A: Unmodified natural	Class A: Unmodified natural
Category B: Largely natural	Class B: Largely natural
Category C: Moderately modified	Class C: Moderately modified
Category D: Largely modified	Class D: Largely modified

Not acceptable:

Category E: Seriously modified	Category E: Seriously modified
Category F: Critically modified	Category F: Critically modified

Figure 7.1: Procedure followed in the determination of the AEMC. →: indicates relationship, ⟶ : indicates possible direction of desirable change (DWAF, 1999).

*Insert page 107

A desktop RDM determination has already been performed for the Luvuvhu River (see van Niekerk pers comm, 2000). Based on expert opinion, the EISC (ecological importance and sensitivity category), DEMC (default ecological management class), PESC (present ecological status category), and Best AEMC (attainable ecological management class) have been determined for different quaternaries (refer to Figure 7.2) in the Luvuvhu catchment area, and are indicated in Table 7.2.

Table 7.2: Luvuvhu River classification (van Niekerk pers comm, 2000).

Quaternary	EISC	DEMC	PESC	BEST AEMC
A91C	Moderate	CLASS C	CLASS D	CLASS C
A91F	High	CLASS B	CLASS D	CLASS B
A91H	High	CLASS B	CLASS C	CLASS B
A91J	Very high	CLASS A	CLASS C	CLASS C
A91K	Very High	CLASS A	CLASS B	CLASS B

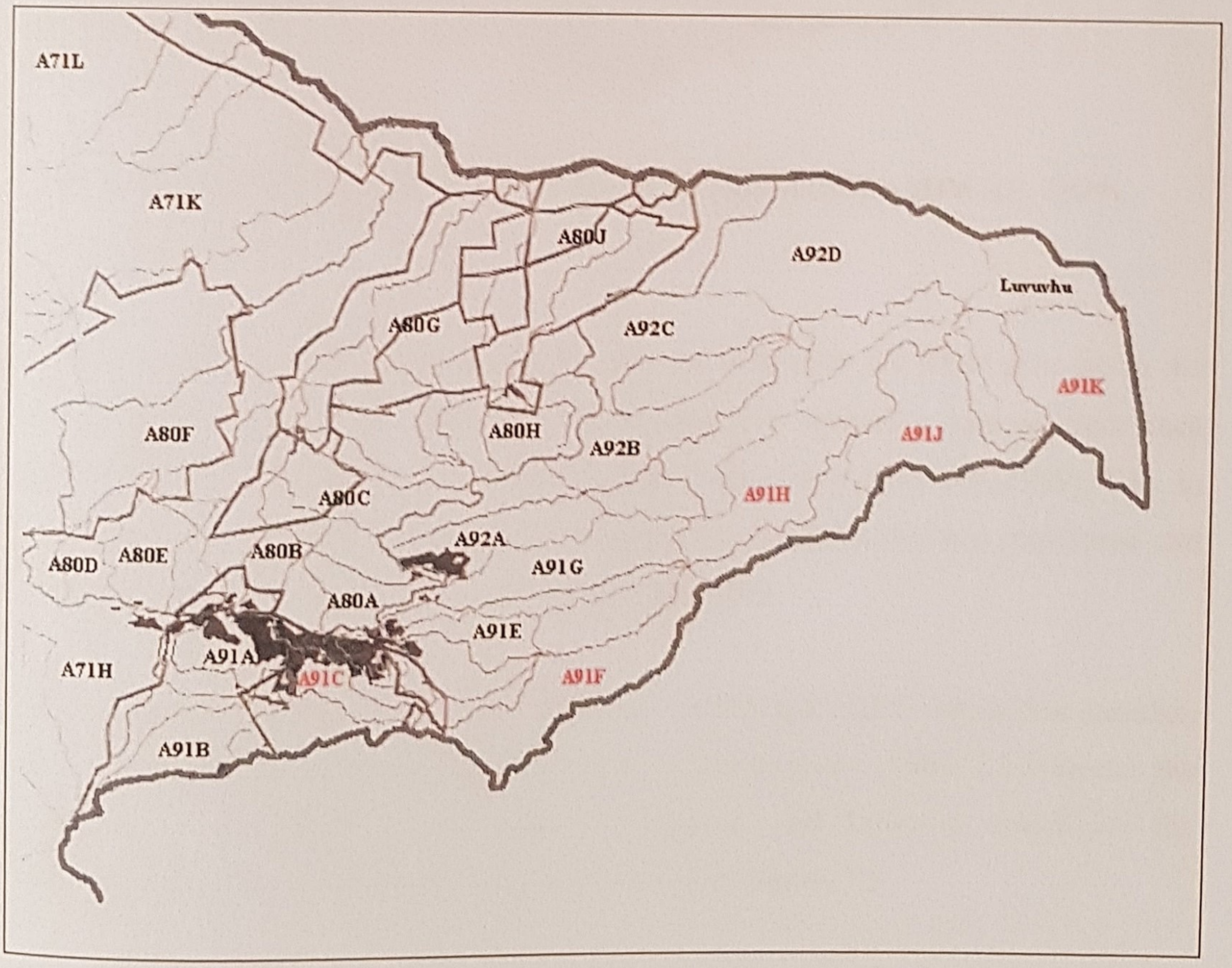

Figure 7.2: Quaternaries in the Luvuvhu catchment area. Quaternaries indicated in red were evaluated during the desktop RDM study.

1. Limpopo plain
2. Central Highlands
3. Bushveld Basin
4. Great Escarpment Mountains
5. Lowveld
6. Lebombo Uplands
7. Highveld
8. Natal Coastal Plain
9. Eastern Uplands
10. Eastern Coastal Belt
11. Southern Coastal Belt
12. Cape Folded Mountains
13. Great Karoo
14. Nama Karoo
15. Western Coastal Belt
16. Namaqua Highlands
17. Southern Kalahari
18. Ghaap Plateau

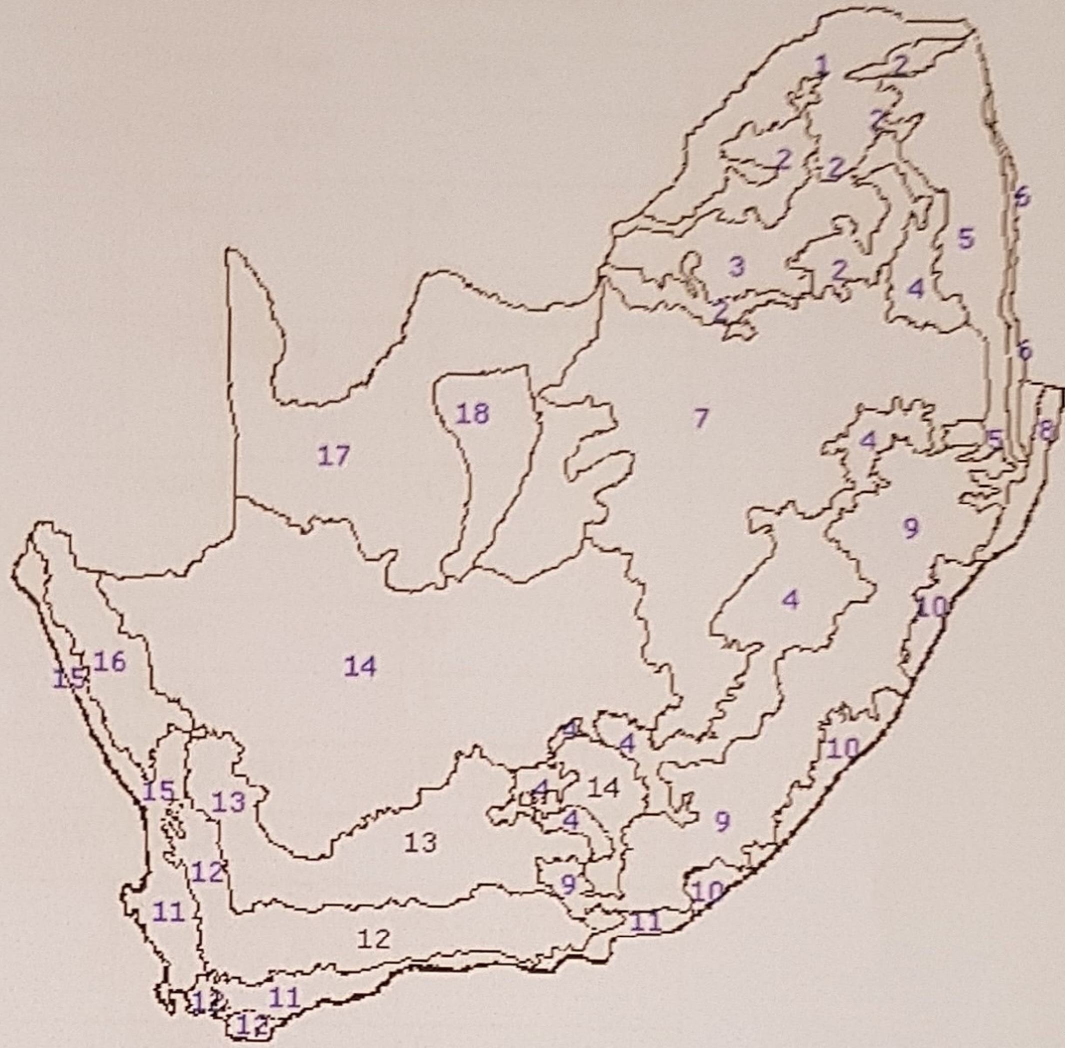

Figure 7.3: Preliminary Level I Ecoregions for South Africa (DWAF, 1999).

SASS4 is one of the biomonitoring tools that are utilised in the RDM process. In the intermediate determination, the invertebrate specialist collects a sample from each habitat and identifies the invertebrates to SASS (family) level (DWAF, 1999). This, in effect, is what was done during this project, where the Luvuvhu was monitored and SASS4 data was collected from each sampling locality.

Thirion *et al* (2000) determined the different SASS4 and ASPT values that correlate to the different classes, for each of the individual ecoregions. Table 7.3 indicates the values that were done for the Central Highlands and Lowveld, which are the ecoregions that the Luvuvhu River falls according to Figure 7.3.

Table 7.3: SASS4 and ASPT values for Central Highlands and Lowveld ecoregions as an indication of biotic condition (Thirion *et al*, 2000).

SASS4	ASPT	Condition	Class
CENTRAL HIGHLANDS			
161-170 >170	>7 >6	Excellent	A
121-160 141-170	>7 5-7	Very Good	B
91-120 121-140	<7.5 <7	Good	C
61-90	<6	Fair	D
30-60	Variable	Poor	E
<30	Variable	Very Poor	F
LOWVELD			
141-160 >160	>7 >6	Excellent	A
106-140 106-160 131-160	>7 6-7 5-6	Very Good	B
76-105 106-130	>5 5-6	Good	C
61-75	4-6	Fair	D
30-60	Variable	Poor	E
<30	Variable	Very Poor	F

7.2 MATERIALS AND METHODS

The sampling localities in the Luvuvhu River that were investigated during this study, and that are situated in the quaternaries A91C, A91F, A91H, A91J, and A91K in Figure 7.2, were classified according to values and ecoregions provided in Table 7.3.

This classification serves as part of the intermediate PESC, as the data obtained provides information concerning the current condition of the water quality, water quantity, physical and the biological state, and the habitat, according to the invertebrate abundance and diversity. As stated, the invertebrates' component forms only a part of the intermediate RDM determination, and hence will be referred to as the 'intermediate' PESC.

This 'intermediate' PESC, based on those localities that correspond to the quaternaries used in Table 7.2, was then compared to the predetermined desktop PESC.

7.3 RESULTS

As stated, the SASS4 and ASPT values, obtained at the sampling localities located in the quaternaries used for the desktop RDM determination, produced the 'intermediate' PESC. These results are compared to the desktop PESC in Table 7.4.

Table 7.4: SASS4 scores and ASPTs obtained at the sampling localities; the 'intermediate' PESC; and the desktop PESC.

Ecoregion	Sampling locality no.	Quaternary	SASS4 score	ASPT	'Intermediate' PESC	Desktop PESC
Central Highlands	9	A91C	117	5.51	C	D
	13	A91F	107	6.68	C	D
	*16	A91H	179	6.88	A	C
	17		173	6.92	A	
	18		153	6.95	B	
	20	A91J	174	6.21	A	C
Lowveld	23	A91K	156	7.09	A	B
	24		203	7.52	A	

* Only the scores for channel 1 were used (refer to Chapter 5).

For quaternaries A91H and A91K the overall PESC will be taken as class A.

7.4 DISCUSSION

None of the 'intermediate' PESC classes, determined according to sampling data, were equal to the desktop PESC classes. In all cases the obtained data generated higher PESCs. Aquatic macroinvertebrates thus indicate a better current ecological status than the attributes used to determine the desktop PESC (refer to Table 7.1).

The two different sets of PESC classes ('intermediate' and desktop) are comparable due to the following:

- The desktop PESC is concerned with water quantity (NWBM) and particular emphasis is placed on flow modification.
- The above determined 'intermediate' PESC was based on SASS4 scores, and the presence or absence of different families indicate not only water quality but flow tolerances as well, as many intolerant families also require flowing water conditions (DWAF, 1999).

This is verified by the similarity between the patterns that were displayed by the SASS4 and ASPT values and the changes in the desktop PESC classes. With reference to Table 7.4, the lowest ASPT score was recorded at sampling locality 9, and the lowest SASS4 score at locality 13. As expected, the desktop PESC for the analogous quaternaries A91C and A91F is set at the lowest class out of all the quaternaries in the Luvuvhu catchment, namely at CLASS D.

Further, for the next two quaternaries (A91H and A91J) the desktop PESC was determined to be higher, and set at CLASS C. This is in agreement with the increase in SASS4 scores that were found at the sampling localities (16 and 20) in these quaternaries.

Finally, the highest ASPTs were obtained at sampling locality. 23 and 24, with the latter also having the highest SASS4 score. These localities are found in quaternary A91K. This quaternary had its desktop PESC set at the highest class (CLASS B) for the whole of the river.

It is however not possible to state which of the SASS4 scores or ASPTs agreed more with the desktop classification of the present ecological status of the quaternaries, without further scrutinising the data in Table 7.4.

7.5 CONCLUSIONS

The RDM process endeavours to protect the ecological integrity of the aquatic environment. Based on the comparison of the desktop PESC and the SASS4 data, SASS4 can be used with a variety of other environmental assessment tools to determine the ecological status of a river, as the SASS4 and ASPT values exhibit a distribution that is comparable to that of the desktop PESC classes.

Although the 'intermediate' PESC classes indicated a higher degree of ecological integrity in each of the catchments evaluated for the desktop study, it should be remembered that other factors, for example biotic factors such as fish and riparian vegetation, are evaluated during an intermediate RDM determination. There is a good chance of the 'intermediate' PESC classes decreasing once all the relevant components have been evaluated. This is perhaps an indication that there should be improved mitigation efforts relating to the circumstances surrounding and impacting on the fish communities and riparian flora.

It can be concluded that ultimately the SASS4 information gathered during this study confirmed the reliability and accuracy of the existing desktop PESC for the Luvuvhu River.

This section further justifies the inclusion of SASS4 as a tool with which to determine intermediate and comprehensive PESCs. SASS4 contributes to the derivation of a higher confidence PESC, as highlighted by the close similarities indicated between the 'intermediate' and desktop PESCs.

In addition, the results of this part of the study prove the RDM methodology on a level with increased certainty and resolution. It lends to the establishment of the new environmental management tool that is the Resource Directed Measures.

CHAPTER 8

INCORPORATING SASS4 INTO THE ECOLOGICAL RISK ASSESSMENT PROCESS

8.1 INTRODUCTION

The goal of the Resource Directed Measures for Protection of Water Resources (DWAF, 1999) is to manage and regulate pollution and land use impacts on the water environment, so as to protect water resource quality. This goal is to be achieved through generating Resource Quality Objectives (RQO). Resource quality objectives (RQOs) for a water resource are set on the basis of acceptable risk: that is, the less risk we are prepared to accept of damaging the Resource Base and possibly losing the services provided by the water resource, the more stringent would be the objectives (DWAF, 1999). 'Acceptable risk' is better described by the 'acceptable probability of damage to the ecological integrity', as the acceptable risk to a certain water resource can be regarded as dissimilar between different water users (e.g. recreational vs. industrial users). South African river systems will thus be categorized according to acceptable levels of human use and impact, and the risk of degradation that a particular use involves (Palmer, 1999).

According to Thornton and Paulsen (1998), assessing changes in the assessment endpoints and reaching conclusions on ecological adversity or significance is ultimately founded on information obtained through monitoring and measurements on ecological systems. Biomonitoring of freshwater ecosystems serves as a valuable tool to gain knowledge of the adverse effects experienced by the assessment endpoints. This is verified by Karr and Chu (1997), who state that the goal of biological monitoring is to "track, evaluate, and communicate the condition of biological systems, and the consequences to those systems of human activities", thus identifying ecological risks.

A primary function of SASS4 is to evaluate the water quality and current biotic condition of a river or stream, based on the diversity and composition of the invertebrate community. This information may then serve as baseline data, with which future scores can be compared to determine the occurrence of an impact on the aquatic ecosystem. A decreased SASS4 score at a particular sampling locality can be indicative of the exertion of stress, mainly chemical (e.g. a toxic effluent) or physical (e.g. habitat degradation), on the invertebrate community. SASS4 thus indicates where impacts have already occurred. According to Suter (1993), a risk assessment that is performed from observation of effects in the field, such as with SASS4, is termed an effects-driven retrospective assessment. A retrospective risk assessment is defined by the USEPA (1998) as an evaluation of the causal linkages between observed ecological effects and stressor(s) in the environment. The risk assessor thus aims to deduce relationships between effects that have already occurred in nature and potential stressors, by investigating possible exposure from various stressors via different pathways and from different media.

Suter (1993) suggests a paradigm for a retrospective risk assessment. It starts off with 'hazard definition', which includes defining the motive for the assessment, describing the environment that is to be assessed, and choosing endpoints. A retrospective assessment may be source-, exposure-, or effects-driven. The paradigm continues with 'measurement and estimation'. Here the available information is used to make inferences about the unknown parts in the stressor-receptor relationship puzzle. 'Risk characterization' and 'risk management' are the two final stages in this framework.

This part of the study is aimed at incorporating SASS4 into the Ecological Risk Assessment process, as this biomonitoring method is already well-established in national biomonitoring programs and has acquired the confidence of local environmentalists. This will facilitate the introduction of risk into legislative and regulatory frameworks in South Africa. It is hence the aim of this chapter to determine how and where SASS4 fits into the Ecological Risk Assessment framework. As habitat assessment plays an important part in the evaluation of SASS4 data, the Integrated Habitat Assessment System (IHAS) will accompany the aforementioned technique in its integration process.

8.2 THE ECOLOGICAL RISK ASSESSMENT PROCESS

According to USEPA (1998), an ERA consists of three primary phases: problem formulation, analysis, and risk characterisation. Murray and Claassen (1999) interpreted and evaluated the USEPAs ERA process and adapted the guidelines to facilitate environmental management in South Africa. Although certain changes have been made to the ERA process, the main objectives and tasks remain the same. If an ERA was to be performed based on SASS4 results indicating the occurrence of an impact, SASS4 and IHAS information would occupy certain stages in the ERA process.

In the following sections there will be looked at where and how SASS4, with an applicable habitat assessment method (i.e. IHAS), will fit into the ERA process proposed by Murray and Claassen (1999). Each phase of the ERA process will be demonstrated with a diagram (refer to figures 8.1, 8.2, 8.4, 8.5, and 8.6), illustrating the associated tasks, while each applicable subsection will be augmented with a selected case study. Sampling locality 13 (Nandoni) in the Luvuvhu River will serve as the case study, as it was deliberated that intrinsic aspects of this site will support the institution of the ERA concept. This locality, as shown in Table 7.4, falls under quaternary A91F. The management class (i.e. best attainable ecological management class) for A91F is set at class B (Table 7.2). According to Table 7.3, for a class B river in the Central Highlands the SASS4 score should be 121-160 with the ASPT below 7, or 141-170 with an ASPT of 5-7. The SASS4 score (107) of locality 13 is thus too low and requires improvement.

8.2.1 Agree on Objectives

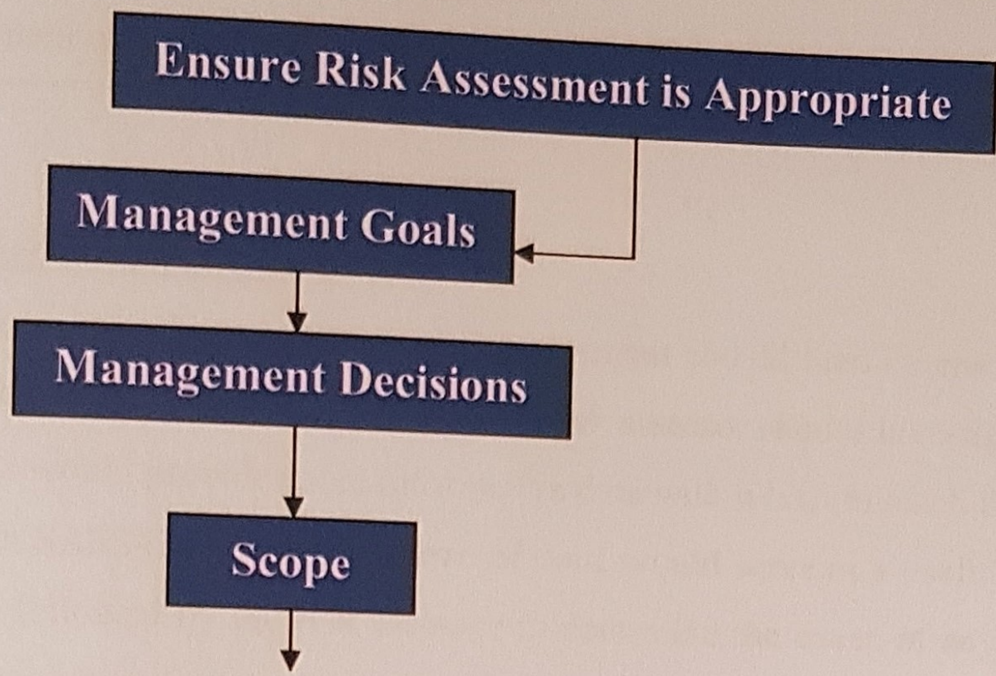

Figure 8.1: Framework of Planning Phase.

Agree on management goals

During the planning of the risk assessment the management goal could be "*for a class B river the SASS4 scores and ASPTs should reflect the biological integrity associated with this management class*". This will keep the assessment process focused on the ecological entity that is to be protected namely the invertebrates.

Case study:
• Improve existing SASS4 score (107) to comply with the prerequisites of a class B river.

Define management decisions

Following this, the management decisions, which determine the means by which management goals can be achieved, are outlined.

Case study:

- Restoration of aquatic invertebrate habitat
- Improvement of the water quality.

Ensure risk assessment is appropriate

The appropriateness of an ecological risk assessment should then be ensured. Before preceding with an ERA the risk manager and risk assessor should investigate whether other methods would provide more informative data with which to reach the set goals. An ERA, with SASS4 as the main source of data, would serve as a useful framework that could be followed by the risk assessor to determine the cause of any changes in the invertebrate community.

Case study:

- An ERA structures the investigation to determine what produced the low SASS4 score.

Scope of the study

Subsequent to the resolution of an ERA's appropriateness, the constraints of the study are established. Temporal, spatial, financial, information accessibility and additional limitations may be encountered.

Case study:

- Data availability: Water quality data was lacking for various sampling localities.
- Temporal: The study only scrutinized information obtained during the three-month sampling period.
- Spatial: The case study is restricted to upstream sites exposed to anthropogenic activities, the environment surrounding sampling locality 13, and the immediate downstream sampling locality.

8.2.2 Formulate Analysis Plan

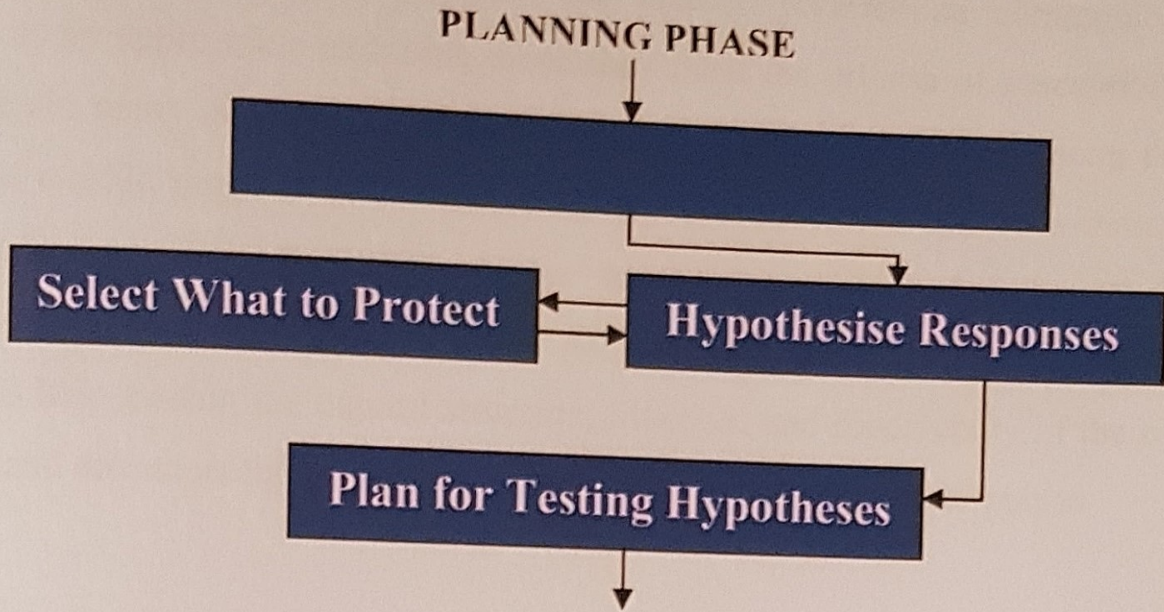

Figure 8.2: Framework of Problem Formulation Phase.

Integrate available information

The problem formulation phase starts off with integrating available information. Integrating the SASS4, IHAS, and chemo-physical data and general site description information would allow the risk assessor to form an idea of the nature of the problem, and to determine if there is a shortage of information/data.

Case study:

- No cased *Trichoptera* and other high scoring taxa (e.g. *Leptophlebiidae*, *Tricorythidae*, and *Aeshnidae*) were found at this site, possibly indicating pollution.
- Recent flood disturbance, surrounding farming activities, and the high conductivity (150 µs/m) measured at this locality (highest out of all the sites - Appendix 5A, Table 5A.5) indicate the presence of the large quantity of sediment.
- The stones out of current biotope is absent.
- There is a dam about 100 m upstream from the sampling locality.

Select what to protect

Next, assessment endpoints are chosen. The benthic invertebrate community is considered as an appropriate endpoint, since they meet the criteria of susceptibility, where there are many different taxa of benthic macroinvertebrates, and among these taxa there is a wide range of sensitivity to all types of pollution and environmental stress. Gentile and Harwell (1998) propose identifying ecological endpoints that reflect either importance to the structure or function of an ecosystem. Aquatic invertebrates help sustain the natural structure, function, and biodiversity of the river ecosystem, and also meet these requirements of suitable assessment endpoints.

Case study:

- Abundance and diversity of the benthic macroinvertebrate community

Hypothesise responses

The problem formulation phase continues with the development of risk hypotheses that forms part of the conceptual model. This includes responses, how exposure occurs and stressor-receptor relationships. Here the integrated SASS4 data can be used to predict possible relationships between invertebrate communities and the estimated stressors. An example (refer to conceptual model diagram in figure 8.3) could be an invertebrate community dominated by functional feeding groups that are associated with eutrophic conditions, namely collectors. If farming activities are present in the area surrounding the sampling locality, then fertilizers (through surface runoff) could possibly be to blame for the organic pollution. Developing risk hypotheses based on SASS4 data is, however, not always such an easy task, as a decrease in a SASS4 score can be attributed to changes in water quality, flow modification, habitat degradation, etc.

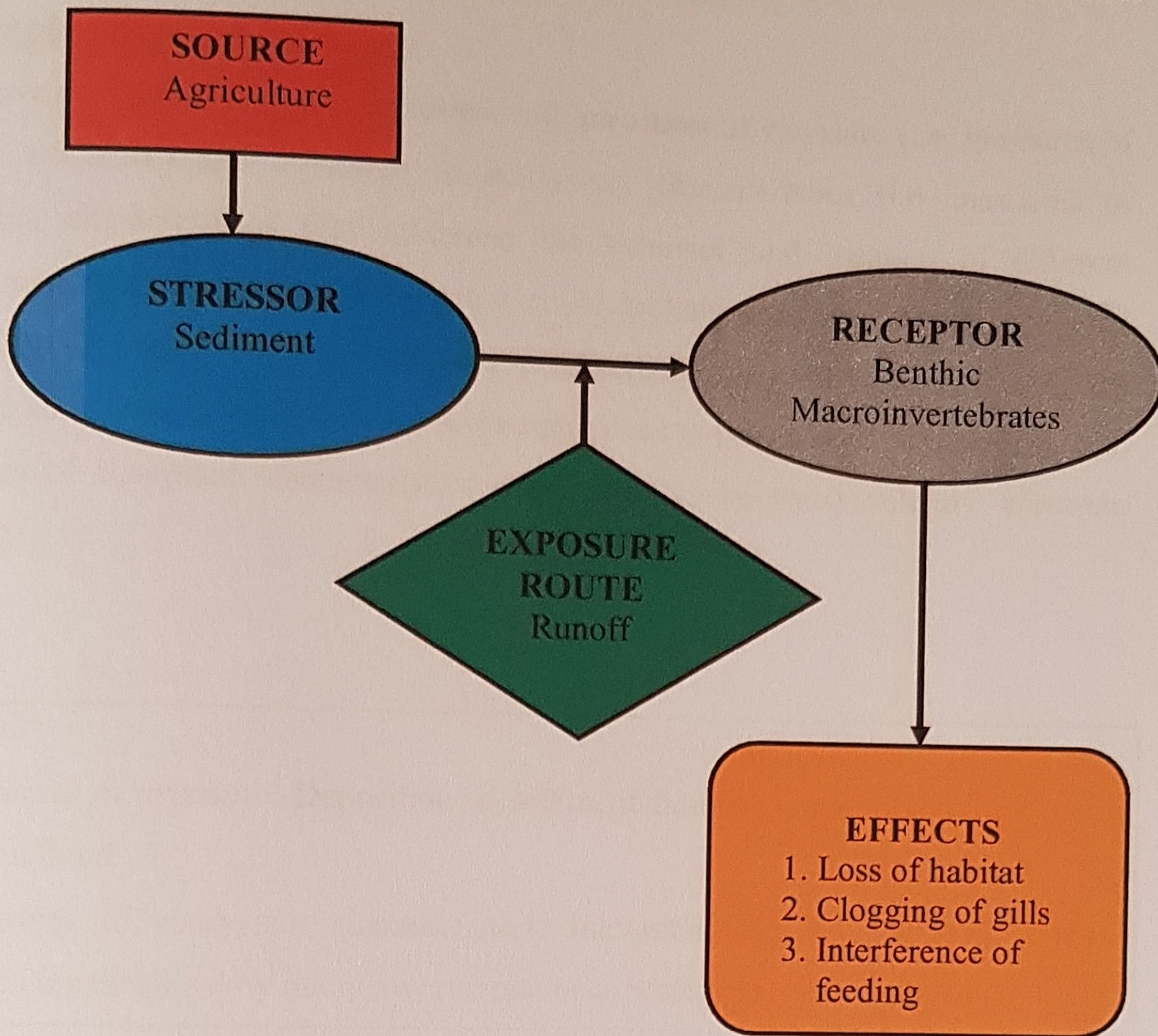

Figure 8.3: Conceptual model diagram.

Case study:

- Based on the management decision, the hypothesis is that the degradation of invertebrate habitat, due to increased sediment, is the main stressor influencing the SASS4 score at this locality.
- The absence of the stones out of current biotope could explain how certain species were not found.
- Although marginal vegetation was sampled, the impoundment of water in the upstream dam could cause low flow, preventing taxa associated with this biotope from being present at this locality.

Produce plan for testing hypotheses

The above risk hypotheses can be tested with measures of exposure (i.e. measures of stressor existence) and measures of ecosystem characteristics (i.e. measures of ecosystem characteristics that influence the behavior and location of different invertebrate taxa). Measures of exposure could include presence of gauging weirs, dams, farming and/or industrial activities, rural settlements, etc. A habitat assessment could be conducted to relate SASS4 score reductions to possible sources or exposure. Measures of ecosystem characteristics could possibly be water velocity, substrate type, etc.

Case study:

- Measures of exposure: Deposition of sediment from agricultural activities and a recent flood.
- Measures of ecosystem characteristics: Interstitial spaces underneath stones, which are inhabited by many invertebrate taxa, were filled with sediment.

8.2.3 Analyse information

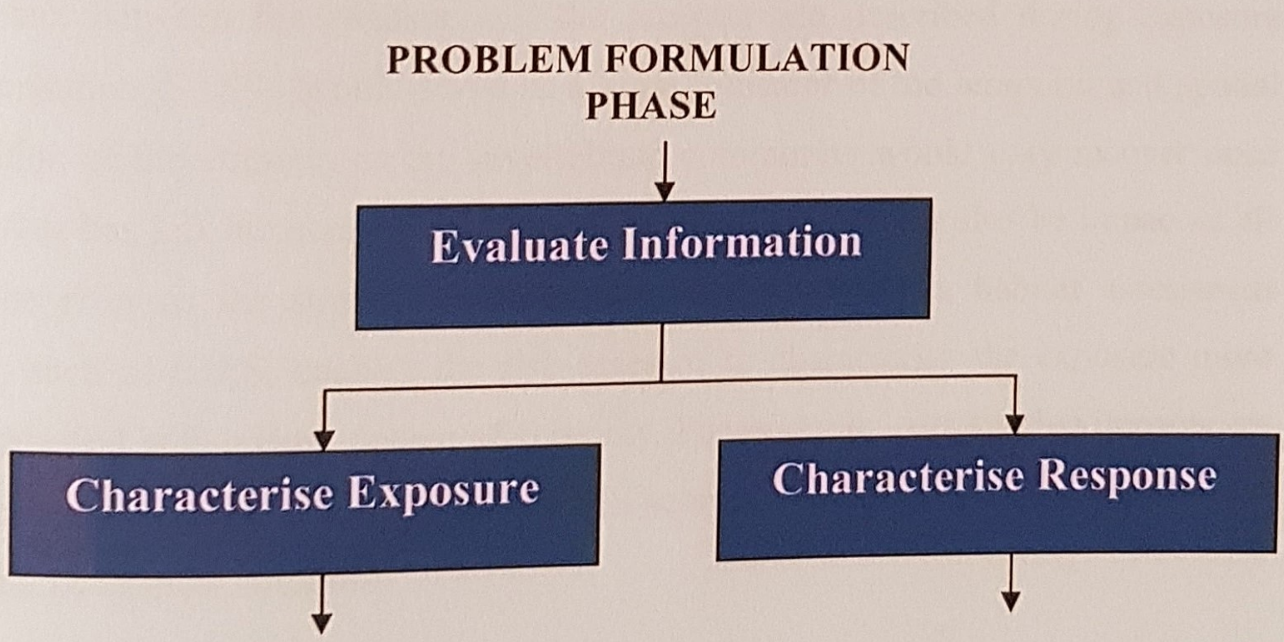

Figure 8.4: Framework of Analysis Phase.

Critically evaluate information

Possible sources of uncertainty should be addressed and evaluated by describing what is known and not known about exposure and effects. As mentioned previously, it is difficult to determine causality when low SASS4 scores are attained, as numerous factors could be to blame. A lack of data, sampling errors, deviation from the SASS4 protocol, and errors in taxa identification could also contribute to uncertainty.

Case study:

- Sources of uncertainty include:

1. Main contributor of sediment, namely agricultural activities or flood.
2. The extent to which the absence of the stones out of current biotope would influence the diversity of the invertebrate community.
3. Deviation from SASS4 protocol.
4. Sampling errors.

Characterise exposure

The source, intensity, and spatial and temporal distribution of the stressor, as well as the contact between the stressor and the receptor are described during exposure characterisation. SASS4 would serve as a good indicator of the temporal and spatial distribution of the stressor, as the invertebrate community would only recover once the stressor has left the ecosystem, and the invertebrates would also be impacted all along the river as the stressor is transported. Incorporating a habitat assessment method, such as IHAS, enables the risk assessor to characterise the exposure more thoroughly and with greater degree of certainty. For example, surrounding impacts are identified during the completion of the IHAS score sheet, which in many cases serve as the source of most stressors.

Case study:

- IHAS indicated recent flood disturbance and surrounding agricultural activities. One or both of these could be the source of the sediment increase at Nandoni.

- The SASS4 score at sampling locality 14, just downstream of locality 13, had improved to 129, with minimal amount of sediment present.

Characterise responses

The most important position that SASS4 information would fill in the ERA framework is during the effects assessment of the analysis phase. It provides valuable information concerning what changes the invertebrate community has undergone in response to a possible stressor, where changes could be a shift in community structure (loss of intolerant taxa; dominance of certain functional feeding group), or the absence of certain taxa, etc. In some cases linking a certain effect to a specific type of stressor, and finding the possible source, is more obvious than for others, e.g.:

- Observed effect = Invertebrate community dominated by *Chironomidae* and *Oligochaeta*

  Possible stressor = Organic pollution

  Possible source = Agricultural activities (fertilizers); rural settlements (sewage), etc.
- Observed effect = Shortage of the Order *Ephemeroptera*

  Possible stressor = Sedimentation (loss of insects' preferred habitat)

  Possible source = Agricultural runoff
- Observed effect = Loss of taxa associated with marginal vegetation

  Possible stressor = Low flow

  Possible source = Impoundments; water abstraction

Once again, it should be stressed that establishing relatively distinct cause-and-effect relationships are difficult, due to the SASS4 scores being dependent on a variety of factors. This task can, however, be assisted through the use of habitat assessment methods (e.g. IHAS, HQI, HAM, etc.), considering that the SASS4 score is influenced to a large extent by the condition of the invertebrates' habitat. Establishing stressor-receptor relationships are further hindered by situations where multiple disturbances are encountered.

Case study:

- The shortage of taxa belonging to the order *Ephemeroptera* and *Odonata* can mainly be attributed to the reduction of interstitial spaces, due to sediment deposition.

8.2.4 Characterise Risk

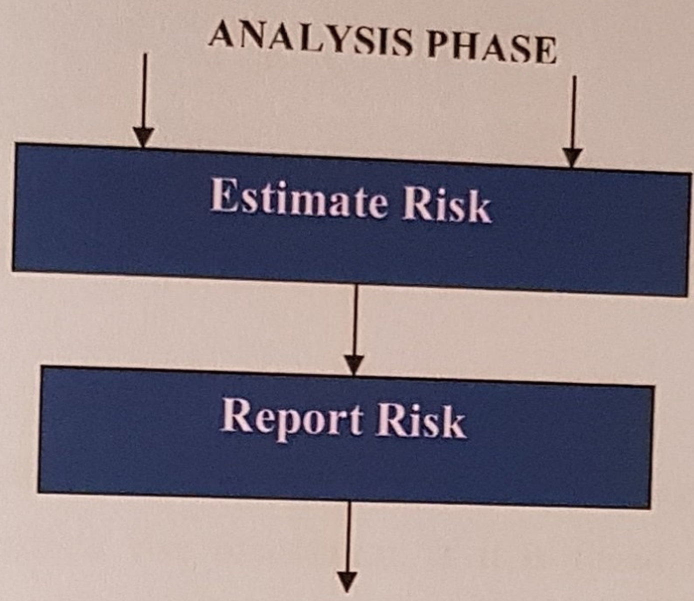

Figure 8.5: Framework of Risk Characterization Phase.

The risk that a certain stressor will induce specific negative effects in an invertebrate community, hence lowering the SASS4 score and disrupting the aquatic environment, could initially be characterized qualitatively, e.g. high, medium, or low. Expert opinion and historical data concerning conditions that are similar to the current situation (if available) would enable qualitative statements to be made about the risk posed.

Case study:

- The probability that the decreased SASS4 score is as a result of sedimentation is high.

8.2.5 Risk Management

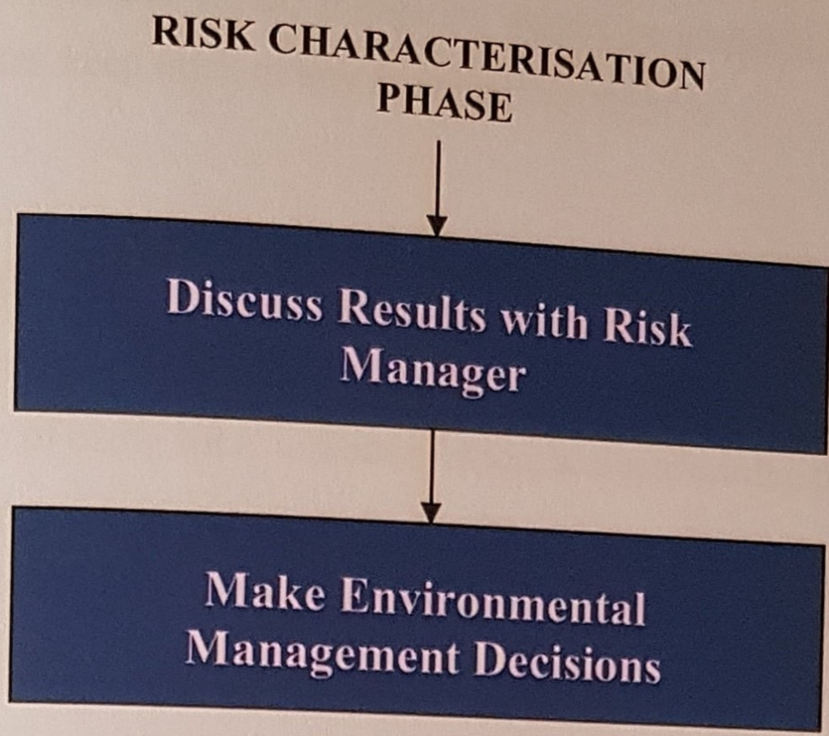

Figure 8.6: Framework of Risk Management Phase.

Finally, the ERA is concluded with risk management. At this stage, the risk manager may require a more detailed risk assessment, if it is found that environmental management decisions cannot be soundly supported by the assessment's results. As ERA is an iterative process, the effects assessment stage could be performed at a more detailed level to deliver sufficient information to facilitate the management of the risk. This could be achieved through the application of a chemical and physical analysis at the sampling locality. Further information could also be gained by performing additional biomonitoring methods (e.g. Fish Community Index, Riparian Vegetation Index, etc.). In the context of the risk assessment framework posed by NSW EPA (1997), SASS4 data could be applied during a primary assessment where it would indicate whether further assessment is required.

Case study:

- If high sediment loads persist, improve land use management in terms of agricultural activities near locality.

Based on the final results of the effects-driven risk assessment, and relevant legal, social, political or economic input, mitigation might be required. Further SASS4 monitoring would then keep track of any remediation progress.

8.3 CONCLUSIONS

The desired outcome of using ERA in South Africa, and performing the process using SASS4 data, is to provide decision-makers with information that must ultimately result in higher efficiency in local environmental management. For example, risk could indicate where water resources are over- or under-protected from certain water uses. It would facilitate predictions of environmental implications of proposals, and could help to identify the least environmentally damaging option.

How will the ERA process have to conform to fit into environmental management conditions in our country? The task faced in introducing ERA to South African circumstances is a daunting one. Unlike the conditions that ERA is normally performed under, here it will be utilized in the assessment of rivers that are mostly situated in areas that are characterized with poverty and a gradual destruction of renewable resources. The process also concentrates more on chemicals present in aquatic ecosystems than on organic enrichment and other disturbances that dominate water pollution in S.A. Fortunately, one of ERA's greatest strengths is that the framework is sufficiently flexible to apply to a broad range of environmental problems (Menzie and Freshman, 1997).

Further, in practice, ecological risk assessment has been the application of the science of ecotoxicology (i.e. the extension of toxicology to the ecological effects of chemicals) to public policy (Suter, 1993). It is thus a strategy for forecasting biological effects (predictive ERA). Biomonitoring methods such as SASS4 and IHAS are means of evaluating the outcome of a proposed project or to observe if an impact has already occurred (retrospective ERA).

Performing a predictive ERA from SASS4 and a habitat assessment method data would mainly be based on guestimates of the extent to which the scores of these biomonitoring techniques would be influenced by a proposed project. Where similar cases exist and the project (e.g. a copper mine) was completed with ensuing impacts on the river ecosystem, extrapolations can be made from scores obtained from surrounding sampling localities to the current situation. In these cases the evaluation of uncertainty would play an important role.

In order for ERA to become established in South Africa, certain points need to be looked at in the near future, namely:

1. The promotion of multidisciplinary research on the theory and practice of risk assessment and management.
2. The use and integration of local monitoring methods into the risk assessment process.
3. The encouragement of intellectual exchange among researchers, risk assessors and policy makers.
4. The use of ERA in legislative, regulatory and other policy considerations.
5. Public involvement in risk assessment and management.

In due course, the amalgamation of ERA and SASS4 will prove beneficial to the protection of our water resources, as SASS4 could indicate the extent of an impact while ERA will provide the means with which to determine causality.

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CHAPTER 9

RECOMMENDATIONS

9.1 RECOMMENDATIONS

1. According to Fairbrother *et al* (1997), effects-initiated assessments (i.e. retrospective assessments) are not risk assessments, as they do not forecast the likelihood of the occurrence of an adverse effect. This statement does warrant some attention being given to the use of the term 'risk' in a retrospective ecological risk assessment. Perhaps an adjustment should be considered in the title of risk assessments that are initiated through observed or known sources, exposure, or effects.

2. During this study, the only chemo-physical variables that were measured included temperature, conductivity and pH. Where the interpretation of the SASS4 data indicated a decrease in water quality, no significant variation was found in these variables. Measuring a wider range of variables, such as dissolved oxygen, and the concentrations of total inorganic nitrogen, total inorganic phosphorus, orthophosphate, ammonia or ammonium, would facilitate linking low SASS4 and ASPT scores to water quality. It should, however, be remembered that SASS4 is a rapid assessment, and serves as a 'red light' to focus attention (in the form of further chemical analysis) on localities where reduced scores are encountered.

3. Only SASS4 and IHAS data for one season was gathered. According to Dallas (1997), the frequency and timing of sampling may influence results and the interpretation thereof, as lotic systems in South Africa exhibit daily and seasonal periodicity, and invertebrate communities exhibit temporal variation (due to life history features and environmental perturbations). To increase the credibility of the obtained information, attention should be given to the temporal variation in SASS4 scores and ASPTs. Sampling should be done for different seasons, and for more than one year.

4. SASS4 and ASPT data were interpreted according to the guidelines set by Chutter (1998) for surface waters that are not naturally acid (pH>6). Only in special and extreme cases were SASS4 scores below 100 and ASPTs below 6. In general, sampling scores were significantly higher. Perhaps a guideline adjustment should be considered, based on a river's circumstances (e.g. parts of river situated in a reserve) and region (e.g. ecoregion). This would make it more possible to notice impacted sites where a set of high scores is obtained.

5. According to Chutter (1998), "the composition of the aquatic macroinvertebrate community is always modified immediately downstream of dams and weirs". This is also often true downstream of bridges. Unfortunately, some of the sampling localities used were indeed sited near weirs and downstream of bridges. There has to be looked at choosing new localities that adhere to the above instructions.

6. It would be advantageous to determine the recovery time of invertebrate communities to different perturbations. This could make it possible to gain knowledge about the time of an impact's occurrence and its severity, based on how long the community takes to return to normal. Recovery time depends on the type of impact and the hydrodynamics of a system (Rosenberg and Resh, 1993). Determining the recovery time from exposure to a certain stressor would be easier for some impacts than for others. For example, flow-induced changes would cause a response in the invertebrate community that would be easier to determine the recovery time from as for instance from exposure to a certain toxic chemical.

7. There is a need to predict the risks posed by future projects to aquatic resources, based on biomonitoring methods. Toxicity tests could be performed on various individual invertebrate organisms with varying tolerances. This would tell us a bit about how the invertebrate community would react if certain tested chemicals were to be released in the aquatic environment. SASS4 could then serve as a means to look retrospectively at the validity of the laboratory devised data. An artificial aquatic environment could also be created under laboratory conditions, containing the most inhabited biotope, namely stones in current. The invertebrates associated with this biotope could then be added. Different impacts could then be

simulated (e.g. sedimentation, variation in flow velocity, adding of toxic chemicals), where after the invertebrate community is sampled, and a SASS4 score generated. These laboratory-based relationships can then be extrapolated to the real environment. Historical data can also be incorporated into a risk assessment to investigate how SASS4 scores and ASPTs reacted to a stressor that is currently under review. These are all possible applications of SASS4 in predictive ERAs, as they enable guesstimates to be made about the risks that particular stressors pose to the invertebrate community structure. Kwiatkowski (1998) states that bioassays (i.e. toxicity tests) and community structure analysis (biological indices) are used in tandem in environmental health assessment to obtain a measure of biological degradation. The feasibility of performing a predictive risk assessment using SASS4 data requires more work in the future.

8. Besides SASS4, data from other biomonitoring tools can be utilised in an ERA. The procedure described in chapter 8 could serve as an example on how to conduct a retrospective ERA based on information gathered from biomonitoring methods, where environmental disturbances are observed. Once again, the available data would be used to postulate causality. These applications offer the opportunity for future investigation.

9. There could also be looked at setting SASS4 scores and ASPTs at certain levels to serve as guidelines for different water uses (e.g. industry, agriculture, domestic, recreation, etc.). If SASS4 scores were then to be reduced, and the environment degraded, remediation would be required to return the invertebrate community to the desired condition.

10. It needs to be ascertained how much the risk of adverse effects will decrease downstream from the point where the disturbance first comes into contact with the waterbody and the receptor (i.e. the invertebrate community). Investigating the exposure (extent, spatial and temporal characteristics of the stressor) and the pattern in the SASS4 scores, and knowledge of the assimilative capacity of the receiving water environment, would assist in performing this task. The assimilative capacity of the receiving water is defined as the capacity of a waterbody to assimilate, through processes such as dilution, dispersion, and

chemical and biologi[...] to a waterbody without water quality changing to the extent that the "fitness for use" of the water or health of the natural aquatic environment is impaired (Roux *et al*, 1999).

11. To increase reliability and acceptance, it is desirable to characterise risk quantitatively. In time to come, existing statistical methods that would allow the quantification of risk based on SASS4 data, should be studied.

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