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RISK AS A TOOL IN WATER RESOURCE MANAGEMENT.

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Title: **Risk as a tool in water resource management**

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Synopsis

The National Water Act (Act 36 of 1998) (NWA) of South Africa makes provision for a quantity and quality of water to be set aside as a Reserve for the provision of basic human needs and for the protection of the aquatic ecosystem for sustainable development of the water resource. An ecological risk approach to water management with a view to the Reserve based *inter alia* on the following:

- Ecological risk is explicitly effect oriented.
- A risk approach will not only address the stochastic characteristic of the ecosystem, but it will also provide a useful tool to address the potential conflict between user and legislator. A risk approach is explicitly effect oriented.
- The probability component of risk supplies a way to bring diverse stressors to a common basis and address the diverse-stressor-multiple source problem.

This study aims to provide a tool to apportion the ecological effect impact attenuation rationally among users.

In order to accomplish this, attention was given to the following:

1. The end-point required by the NWA must be related to end-points at lower organisational levels of the ecosystem. A model is proposed to do this based on the logical relationship between ecological phenomena. Although there is a dearth of information to use in the model, it may contribute to the characterisation of uncertainty with this type of projection.

2. The mathematical formulation of the ERA process has apparently not received much attention in the technical literature. A mathematical formulation of the risk of a single stressor is proposed in both probability and fuzzy logic terms. The risk is expressed as the conjunction of the likelihood of effect conditioned on the stressor occurrence and a likelihood of stressor occurrence.

3. When diverse stressors occur together and no other information is available on their interactions, the aggregate stressor risk may be expressed as the disjunction of individual stressor risks. The value of this approach is investigated in some hypothetical but realistic case studies.

4. The problem of apportionment of impact attenuation burden among multiple dischargers of diverse stressors is similar to waste-load allocation (WLA). Obtaining an equitable distribution of the effect attenuation burden that recognises the technological and economic limitations in a catchment, is an optimisation problem. The diverse-stressor-multiple-source problem is first formulated as a fuzzy optimisation problem, which is solved using a genetic algorithm. This approach is investigated in a hypothetical (but possibly realistic) case study. The objective of the optimisation is the maximisation of the acceptability of the regulated situation. For the regulator

this is assumed to mean the minimisation of ecological risk, while for the stressor source manager this might be influenced by technological and economic considerations. The degree of attenuation of the stressor is chosen as the control variable.

Key terms: Ecological risk; Probabilistic risk; water quality management; fuzzy logic; fuzzy risk; optimisation; Water Act.; Resource management.

Samevatting

Die Nasionale Waterwet (Wet 36 van 1998) (NWW) bepaal dat 'n bepaalde hoeveelheid en gehalte water opsy gesit word as 'n Reserwe vir basiese menslike gebruik sowel as vir die beskerming van die akwatiese ekostelsel. Daarbenewens, word die verpligting op die staat geplaas om die waterhulpbron volhoubaar te ontwikkel. Die ontginning van die hulpbron sal kennelik druk plaas op die akwatiese ekostelsel. 'n Ekologiese risiko benadering in hulpbronbestuur word voorgestel, ondermeer omdat:

- Ekologiese risiko is eksplisiet effek georiënteerd.
- 'n Risko benadering tot hulpbronbestuur sal nie net die stogastisiteit en onsekerheid wat die ekostelsel kenmerk, kan aanspreek nie, maar voorsien ook 'n veelsydige stuk gereedskap wat gebruik kan word om die potensiële konflik tussen gebruiker en beskermer aan te spreek.
- Die waarskynlikheidskomponent van risiko bied 'n manier om diverse stressors op 'n gemeenskaplike basis te plaas om die diverse-stressor-veelvuldige-bron probleem aan te spreek, d.w.s. dié probleem waar diverse stressors wat in verskillende eenhede uitgedruk word maar tot dieselfde globale effek bydra en daarbenewens nog uit verskillende bronne kom, te bestuur.

Hierdie studie poog om die gereedskap te ontwikkel wat die ekologiese impakbekampingslas op 'n rasionele basis tussen gebruikers toe deel.

Ten einde hierdie doel te bereik word aandag gegee aan die volgende aspekte:

1. Die eindpunt (tw. volhoubaarheid) wat deur die NWW vereis word moet in verband gebring word met eindpunte by laer organisasie vlakke van die ekostelsel. Hiervoor word 'n model voorgestel wat gebaseer is op die logiese verband tussen ekologiese verskynsels. Hoewel besonderhede vir die model skaars is, kan dit bydra tot die uitspel van onsekerheid by hierdie vorm van eindpunt projeksie.

2. Die wiskundige formulering van ERA het min aandag in die vakliteratuur gekry. 'n Wiskundige uitdrukking van risiko skatting vir 'n enkele stressor word voorgestel in beide waarskynlikheidsleer formulering en newellogika (Eng. "fuzzy logic") formulering. Die risiko vir 'n stressor word uitgedruk as die konjunktiewe samestelling van die verwagting van effek gekondisioneer op die stressor voorkoms en die verwagting van die stressor voorkoms.

3. Wanneer diverse stressors saam voorkom, en geen verdere inligting beskikbaar is oor hulle wisselwerking nie, word die gesamentlike risiko voorgestel as die konjunktiewe samestelling van die afsonderlike risiko's. Die waarde van hierdie benadering word getoon aan die hand van hipotetiese maar realistiese gevalle studies.

4. Die probleem van toebedeling van impakbekampingslas tussen veelvuldige stressorbronne is soortgelyk aan die afval-beladingstoebedeling ("waste load allocation") probleem. Om 'n

eweredige effekbekampingslas te verkry wat die ekonomiese en tegnologiese beperkings van verkillende watergebruikers in die opvangebied in aanmerking neem, is 'n optimiseringsprobleem. Die diverse-stressor-veelvuldige-bron probleem word eers as 'n newel optimiseringsprobleem geformuleer wat dan met behulp van 'n genetiese algoritme opgelos word. Die benadering word aan die hand van 'n hipotetiese (maar moontlik realistiese) gevallestudie ondersoek. Die doelwit van die optimisering is die maksimisering van die aanvaarbaarheid van die gereguleerde situasie. Vir die wetstoepasser is die beperkings van ekologiese risiko waarskynlik belangrik terwyl koste en tegnologiese faktore waarskynlik vir die stressor bestuurder belangrik is. Die graad van stressor vermindering is as beheerveranderlike gekies.

Sleutelterme: Ekologiese risiko; waarskynlikheidsrisiko; watergehaltebestuur; newellogika; newelrisiko; optimisering; waterwet.; hulpbronbestuur

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CHAPTER 2: BACKGROUND AND THEORETICAL CONSIDERATIONS

References

Part 3: Scientific Papers 183

Paper 1: Jooste S and Claassen M (2001) Rationale for an ecological risk approach for South African water resource management. *Water SA* , **27**(3), 283-292

Paper 2: Jooste S (2000) A model to estimate the total ecological risk in the management of water resources subject to multiple stressors. *Water SA*, **26** (2), 159-166.

Paper 3: Jooste S (2001) A possibilistic approach to diverse-stressor ecological risk estimation. *Water SA*, **27** (3), 293-302

Paper 4: Jooste S (2001) Ecological concern as a factor in the optimal attenuation of diverse stressor sources in a stream. *Wat. Sci. Technol.* **43**(7), 239 - 246.

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Abbreviations

Λ or L:	A likelihood measure (such as probability, possibility or necessity)
AEL:	Acceptable effect level
AEV:	Acute effect value (from the SAWQG)
ASL:	Acceptable stressor level
BCF:	Bio concentration factor
BEL:	Benchmark effect level
CAP:	Continuous assessment paradigm (see Appendix Chapter 1)
DO:	Dissolved oxygen
DSMS:	Diverse-stressor multiple-source
EQO:	Environmental quality objective
ERA:	Ecological risk assessment
ERBM:	Ecological risk-based management
ESL:	Expected stressor level
GA:	Genetic algorithm (for optimisation)
Inf:	Infimum (lowest lower bound)
LBB:	Lethal body burden
LC50:	Median lethal concentration
Max:	Maximum
Min:	Minimum
MOA:	Mode of action
MOOP:	Multiple objective optimisation problem
NOEC:	No observed/observable effect concentration
NWA:	National Water Act (Act 36 of 1998)
QAP:	Quantal assessment paradigm (see Appendix Chapter 1)
RDM:	Resource directed measure (provided for in the National Water Act)
RO:	Risk objective
SAWQG:	South African Water Quality Guidelines (1996 edition)
SDC:	Source directed control (provided for in the National Water Act)
SRR:	Stressor response relationship
Sup:	Supremum (highest upper bound)
WET:	Whole effluent toxicity
WLA:	Waste-load allocation

Definitions

{} denotes a set of discreet values, [ ] denotes a continuous interval, *sup*{...} is the highest upper boundary of the set, and *inf*{...} denotes the lowest lower boundary of the set.

Biodiversity: "The variety of life at all levels of organization, represented by the number and relative frequency of items (genes, organisms and ecosystems)"(USEPA, 1997a).

Degree of membership (μ)*:* The Zadehian view: The degree of membership of a value x to fuzzy set A $\mu_A(x)$ is a function which describes the congruence of the perception of x the qualification(s) of A (it expresses the "A-ness of x"). This view supposes that the datum is vague and therefore that μ is the extent to which an observation agrees with the vague concept. The epistemic view (Kruse, *et al.*, 1994): μ is a probability distribution of how well an observation coincides with a specific datum which is only known with uncertainty. It differs from probability in that (*inter alia*) while probabilities sum to 1, in general, membership functions do not.

Ecological risk assessment (ERA): the technique that "evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors"(EPA, 1996). In practice it is the application of the science of ecotoxicology to public policy (Suter, 1993).

Epistemic: Dealing with the nature of knowledge and understanding.

Fuzzy logic: A branch of logic that deals with an infinite number of truth values. If x represents the truth value of a statement, then in Boolean logic $x \in \{0,1\}$ while in fuzzy logic $x \in [0,1]$.

Hazard: The potential of a substance or situation to cause harm.

Integrity: "The state of being unimpaired, sound" (DeLeo and Levin, 1997), "the quality or condition of being whole, complete". The functional definitions are more diverse: "the interaction of the physical, chemical and biological elements of an ecosystem in a manner that ensures the long term health and sustainability of the ecosystem" (USEPA, 1997a), or "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a full range of elements (genes, species and assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region" (Karr, 1996). Other definitions appear to be subsets of these definitions (Cairns, 1977, Karr and Dudley, 1981, Noss, 1990, Rapport *et al.*, 1996).

Likelihood: An expression of the sense of expectation of an observer about an event whether based on repeated observation of identical or morphologically similar events. Can be expressed in terms of probability or possibility (fuzzy) theoretical terms.

Necessity measure: The necessity measure $Nec_\pi(A) = inf\{1 - \pi(\omega) \mid \omega \in \Omega \backslash A\} \in [0,1]$. The necessity measure is related to the possibility that the uncertain event ω0 belongs to the universal set Ω without the set A and is therefore a stonger measure indicating that $\omega \in A$ than the possibility measure.

P(A /B) : The probability of A conditional on B.

P(AB) or P(A∧B): The probability of A and B; or the probability of A in conjunction with B.

Phenomenon: That which appears real to the senses regardless of whether the underlying existence is proved or its nature understood.

Possibility measure *:* A measure of the possibility that an event may occur. The possibility measure for event A, $\Pi_\pi(A) = sup\{\pi(\omega) \mid \omega \in A\} \in [0,1]$. If the possibility of an event is 1 it is entirely possible, while 0 indicates that the event is not possible. The possibility measure does not give any indication of the probability of an event.

Resilience: "The ability of an ecosystem to adapt to change (or stress)" (USEPA, 1997a), or, "the ability to maintain integrity when subject to disturbance" (Holling 1973).

***Risk*:** "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1993) or the probability of observing a specified (unacceptable) effect as a result of a toxic chemical exposure (Bartell, *et al*, 1992). In essence, whether explicitly or implicitly, risk contains elements of: a) likelihood, b) target and c) unacceptable effect. The manner in which the likelihood is expressed introduces gradations to the concept: when a situation allows for Aristotelian (binary) logic and likelihood can be expressed as a probability, then the common form of risk assessment is recovered. However, when fuzzy logic is required and likelihood is expressed in possibilistic terms then fuzzy risk assessment is called for.

Sustainability **:** "the ability of an ecosystem to support itself despite continued harvest, removal, or loss of some sort" (USEPA, 1997a). Implicit in this definition is the assumption that sustainability is time and stressor dependent.

t-norm and t-conorm: Used to define generalised intersection and union operators respectively for fuzzy sets.

Truth value*:* The truth value of a proposition is the degree to which the content of the proposition agrees with the assessors perception of reality. The truth value can be calculated as the compatibility of the possibility distribution representing the proposition with the possibility distribution representing the state of knowledge (Du Bois and Prade, 1988, p126)

Structure

This document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

Part 3: Presents some of the papers that have been published in peer reviewed literature and that are included for quick reference.

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Part 1:

Overview

PART 1: OVERVIEW

1. BACKGROUND

This study originated from the thinking around the South African National Water Act (NWA) (Act 36 of 1998) which replaced an older Act dating from 1956. Three aspects of the NWA had a particular impact on this study:

The NWA guarantees only two rights: sufficient quantity and quality of water to supply basic human needs and to ensure the sustainable functioning the aquatic ecosystem. This quantity and quality constitutes a Reserve, which needs to be protected.

It makes provision for measures to protect the resource as well as to control sources of pollutants (or stressors).

It makes provision for a classification system for resources.

> An ecological **stressor** could be any **substance**, group of substances, a **flow-related** quantity, an in-stream- or riparian **habitat condition** or presence of biota that is **not normally expected** at a given time and place

This study deals particularly with the ecological requirements; briefly referred to as the "ecological reserve". (For more detail on the NWA and its requirements see Part 2, Chapter 1.)

The concept of an ecological reserve developed from the notion that ecosystems are generally fairly resilient and if they are not "pushed too far", they can usually regain the level of services practically indistinguishable from the pre-impact level. It was reasoned, however, that there may be a point at which the system is "pushed too far" so that it then "crashes". A "crashed" system would of course be undesirable, but exactly what constitutes that "crash-point" is uncertain. All that seems reasonable to assert is that the more the system is "pushed" (in the sense of moved away from pristine condition), the greater the likelihood the system will "crash". So, in a broad and as yet undefined sense, the further the system moves from its pristine state the higher the risk of system "crash". From these vague roots the concept of "risk" and particularly "ecological risk" intuitively appeared to be useful. The resulting "grey scale" of risk can be discretised to serve as the basis for a classification system for resources where one end of the scale represent insignificant risk while the other represents unacceptable risk.

This study proposes the use of ecological risk as a decision support tool in water resource management in support of the protection of the ecological reserve. "Ecological risk" and "ecological risk assessment" have become fairly well established as a decision support tool in environmental management as is shown by the literature cited in Parts 2 and 3. The terms "risk" and "risk assessment" have come to take on a wide variety of meanings and encompass a wide variety of practices. This study attempts to find a suitable expression of risk and examines some theoretical concepts around its application to water resource management.

Ecological risk assessment (ERA) for the aquatic environment under the NWA should estimate the likelihood that loss of sustainability will result from the occurrence of aquatic stressors

This study lays no claim to providing new insights into ecological mechanisms that are involved in vague terms like "system crash", "pushed too far". It accepted that there are experts in biology and ecology who can produce elegant, precise and scholarly definitions for these vague terms. As a point of departure, these are used in a phenomenological sense, i.e. without knowing the biological and ecological mechanisms, "pushed" simply refers to the phenomenon "inducing a movement away from" and "crashed" simple refers to a phenomenon "not being able to produce what is expected". So, where some more precise terminology is used, it must be accepted that these are from a relative layman's point of view. It is hoped that where more precise information becomes available, it will still be useful within the theoretical framework provided here with some adaptation of the methodology.

2. GOALS

In this study three main issues are addressed:

1. The rationale for the use of ecological risk - Is risk really conceptually useful in water resource management with the aim to ensure sustainability?
2. Is there a mathematical construct that could be used for risk calculation in ecological risk assessment in the NWA context?
3. How could risk be applied in a multiple stressor multiple source environment?

3. RATIONALE FOR THE USE OF ECOLOGICAL RISK

"No, no!", said the Queen. "Sentence first - verdict afterwards"
– Lewis Carroll, Alice in Wonderland

> Decisions regarding water quantity and quality often have to be made based on meagre information, the impact of which may either justify or condemn the decision.

The unenviable task of the water resource manager may at times seems to call for the reasoning of the queen during the trial in *Alice in Wonderland.* Decisions regarding water quantity and quality often have to be made based on meagre information, the impact of which may either justify or condemn the decision. The reason for this is rooted both in the characteristics of the aquatic ecosystem and our knowledge and use of it. This section addresses the first goal of the study.

3.1 SOME FUNDAMENTAL ISSUES

The event referred to as "ecosystem crash" is a manifestation of impact on the specific assemblage of aquatic organisms making up that ecosystem. The identity of the organisms, their interactions and their relative abundances are determined by a number of both biotic and abiotic factors. In the pristine state, these factors are in dynamic equilibrium, identifying the reference condition for describing system integrity. Now three very fundamental assumptions have to be made:

- Pristine, un-impacted ecosystems do not "crash". Even extreme hydrological events such as floods or droughts are part of the natural regime of ecosystems.
- Aquatic organisms would react to a change in the natural state of their physical, chemical, and biological environment.
- This "crash" only takes place when an unnatural condition is imposed on the system, such as by anthropogenic intervention. Deviation from the pristine state of the ecosystem (interpreted as loss of biotic integrity) would increase the likelihood of reaching that "crash point". The pristine state defines the condition of trivial (or *de minimis*) risk while the crash point defines a condition of unacceptable (or *de manifestis*) risk.

> Extreme natural events such as droughts and floods, which are part of the pristine state regime, are not considered as stressors.

So, in principle sustainable ecological water resource management is simple: manage the physical, chemical and biological environment within suitable limits and system "crash" will be avoided. But what are those "suitable limits" providing a suitable margin of safety?

3.2 COMPLICATING FACTORS IN ECOLOGICAL RESERVE MANAGEMENT

Determining the suitable limits for management is complicated by noting that in dealing with the ecological reserve, or any system where ecological sustainability is an issue, scientists and managers have to address:

Vaguely defined systems (see Part 2 Section 2.3.2 and Part 3 Paper 1)
When dealing with the impact of some form of water use on a specific river reach it could be argued on the one hand that the entire globe is one big ecosystem with internal links of different strengths. On the other hand it could be argued that only the individual organisms in that reach and their direct interactions constitute the ecosystem. To a certain extent both are correct. Between these two extremes system boundaries are a matter of opinion. Of course, in each river or stream and in any given reach of that stream the identity of organisms that make up the system would be different, their individual susceptibilities to environmental factors would be different, and their interactions would be different.

Fragmentary knowledge and uncertainty in its interpretation (see Paper 1, Part 3).

While extensive systematic studies have been performed on certain aquatic species, knowledge of the interaction among species and between species and their environment is not always as well developed. While toxicology (the science of the interaction of substances and individual organisms) has developed into a reasonably exact science, the same cannot always be said for ecotoxicology (the science of the interaction between substances and ecosystems). Even where extensive observations of stimuli and their responses are available, the interpretation of the results is not always uniform. Different conceptual approaches to looking at the same set of observations leads to different models of the system under observation. Different models may yield different assessments of future system response. Different assessments may, in turn, lead to different management strategies.

Systems that are subject to various forms of randomness (see Section 2.3 in Part 2and Papers 1 and 2, Part 3).

In contrast to the previous problem that could conceivably be resolved by more intensive study, randomness is not reduced by study. Randomness (or stochasticity) is often an integral part of ecosystem dynamics. Randomness in ecosystem response is also influenced by randomness in the hydrological cycle (e.g. rainfall, run-off etc.) and by individual variability in response to stressors. The problem, of course, usually arises when the mind-set is deterministic.

A variety of different stressors, each of which may to a greater or lesser extent have an impact on the aquatic ecosystem (see Part 2, Chapter 3 and Part 3, Paper 1).

Conventionally, undesirable substances or energy (in the form of heat) added to water were considered important. However, the amount and timing of water supply and in-stream and riparian habitat condition are also important and may, in some cases, even be more important than water quality in determining ecological impact. Each of these is quantified in different units. Each of these may cause "ecosystem crash". How does one decide on the seriousness of the combined impact? In order to facilitate management, it would be useful (if not necessary) to rank these stressors on a common basis.

Ensuring environmental protection while at the same time not stifling progress (see Part 2, Chapter 4 and Part 3, Paper 4).

Theoretically it is simple to take a precautionary approach when dealing with multiple stressors – to select levels of these stressors where there would be no known effect. However, in a developing, water scarce country like South Africa, this is not so easy. There is a significant need for economic upliftment and development in what is otherwise a frail economy. Water treatment facilities range from highly sophisticated to non-existent. In large areas of the country agriculture is dependent on irrigation from surface water resources and dilution capacity is very limited. An entirely precautionary approach in water resource management may, in some areas, have a devastating economic and sociological effect.

All of the above contribute to an unenviable management situation. From the above, it would appear to be practically impossible to define which set (or sets) of values of physical, chemical and biological variables define that "crash point" and without that information it would impossible to define what a safe margin would be. All that can reasonably be assumed is that the likelihood or probability of ecosystem "crash" increases as deviation from pristine levels increases.

3.3 APPRAISAL OF RISK AS RESOURCE MANAGEMENT TOOL

Some of the important and useful characteristics of risk include:

a. Risk makes use of **two important types of information**: What we know about what would happen to a system when it is exposed to a stressor (i.e. an **effect assessment**), and what we know about the stressor's occurrence (i.e. an **occurrence assessment**). The first question is the basis for a hazard assessment. It does not concern itself with how the stressor behaves in the real world. What risk as a methodology does is to bring the stressor occurrence characteristics in as part of the assessment.

b. Ecological risk needs an **end-point**, i.e. a specific expression of what sort of effect is being assessed. In the case of the ecological reserve, the end-point required by the NWA is "loss of sustainability" (that is the "statutory" end-point). This end-point has a specific value for the public. On the other hand, the scientists who have to assess the impact of a stressor usually don't really have any information specifically relating to "loss of sustainability" as such, but they may infer "loss of sustainability" from other information such as "disappearance of a key species" (that is a "surrogate end-point"). Both statutory and surrogate end-points may be subject of debate and/or negotiation. Projecting from the surrogate to the statutory end-point is not trivial (see Part 2, Chapter2 and its Appendix and Part 3, Appendix to Paper1)

c. A particular characteristic of risk (in the technical sense used here) is its **expression in terms of likelihood** (e.g. probability). If the end-points for the assessment of risk resulting from different types of stressors are the same, then likelihood is practically a unitless way of comparing and **expressing the impact of diverse stressors** (see Part 2 Chapter 3 and Part 3, Papers 2 and 3). This is because the likelihood expression is equipped to handle the complicating factors above better than a hazard approach.

Dealing with technical issues in resource management for the protection of the ecological reserve

Issue	How issue can be addressed on a risk basis	Further Information
Uncertainty in models and innate randomness (stochasticity)	Calculation of probabilistic risk. Can be expressed as uncertainty in the calculated risk	Part 2, Chapter 3 and Part 3, Paper 2
Vaguely defined systems and fragmentary knowledge	Possibilistic risk based on fuzzy logic	Part 2, Chapter3 and Part 3, Paper 3
Assessing impact for a diversity in stressors	Risk aggregation	Part 2, Chapter 3 and Part 3, Papers 2 and 3.
Relating the regulatory (statutory) end-point for an assessment the surrogate end-point	Projection model for assessment confidence	Part 2 Chapter2 and Example in Part 3, Paper 1.
Deriving criteria for the management of multiple sources of diverse stressors	Optimisation to risk objectives	Part 2, Chapter 4 and Part 3, Paper 4

d. A risk approach tends to be **less wasteful of available information** than a hazard approach to stressor management. As indicated in a), a hazard approach tends toward focussing on critical effect benchmark values, i.e. stressor levels that represent selected levels of effect that are perceived to be important by role players in the assessment process. How effect-levels change at stressor levels above and below the benchmark is neglected in the assessment. The major effort in a hazard assessment is focussed on how the stressor presents itself. A risk approach has the potential (even if not always used as such) of being able to utilise both types of information. (See Part 2, Appendix 1 for a discussion of the risk and hazard paradigms). In addition, it is a vehicle to expresses some forms of uncertainty and its impact on a situation assessment (see Part 3, Paper 2).

Because of the factors above risk is also a more arduous approach to resource management. The extra effort pays off by providing a very versatile decision support tool. It is possible, for example, to trade off stressors against each other once a risk goal for a resource has been set. This is particularly useful in addressing factor 5 above (the diverse stressor multiple-source problem, see Part 2 Chapter 4 and Part 3 Paper 4).

The likelihood component of risk can be expressed either qualitatively or quantitatively. Expressions of likelihood can be based either on probability theory, which has a strong mathematical and historical underpinning, or it can be based on fuzzy logic, which has an advantage in dealing with vague expressions often encountered in descriptive ecology. The most suitable expression will depend on the application.

3.4 RISK OBJECTIVES

In applying risk in a resource management framework two types of application can be distinguished: using risk merely as a ranking tool, where the actual risk magnitudes do not matter, or, using risk explicitly.

In the latter case it is assumed that risk objectives will be generated. Risk objectives (e.g. the probability of the loss of species should be $< 10^{-4}$) would be analogous to other forms of in-stream objectives, with the exception that they are essentially dimensionless (referring only to an undesired effect, such as loss of sustainability).

4. TECHNICAL ISSUES IN USING RISK

In addressing the complicating factors in resource management in support of the ecological reserve (above) a number of technical issues needed to be addressed.

4.1 DEFINITION OF RISK

A variety of definitions for risk were encountered in environmental risk assessment literature. For the purpose of this study risk was defined as the likelihood that a loss of sustainable ecological function will occur (Part 2, Paper 1).

4.2 ESTIMATION OF RISK

From the discussion of the components of a risk assessment (Part 3, Papers 1 and 2) a risk assessment should combine a likelihood assessment of effect with a likelihood assessment of occurrence. A number of methods were encountered:

Ratio of benchmarks

The Predicted Environmental Concentration to (Predicted) No-Effect Concentration ratio is one example. If the ratio is less than 1 then no risk exists while if larger that 1 a risk exists. This appears to be little more than a hazard assessment in weak disguise.

Probability of effect benchmark

This requires the calculation of the probability that the environmental concentration will be larger than a benchmark concentration. This still does not provide information on what would happen if the concentration is larger than the benchmark concentration.

Degree of overlap

This method involves determining the area of overlap between an effect likelihood curve (expressed as the likelihood of effect vs. stressor level) and the stressor occurrence likelihood curve (like the probability density function of stressor level occurrence). While conceptually simple, it is not quite clear how to interpret the result.

> The event conjunction model is useful for calculating a stressor-specific instantaneous risk. The stressor-specific risk may be calculated from either the maximum instantaneous risk or from the cumulative risk for a specific situation.
>
> The aggregate risk could be estimated from the disjunction of stressor-specific risk.

Occurrence and effect event conjunction

In general the risk assessment literature recognises that risk depends on some form of conditional probability. As far as could be established, this type of formulation does not appear in the ecological risk assessment literature referenced in this study.

From a theoretical perspective it seemed feasible to assert that a risk only exists when two events occur simultaneously: the event that a hazard exists and the event that a stressor occurs. As a corollary to that one might say that a stressor is only defined as such when it can result in the undesired effect that is chosen as the end-point (see Part 2, Chapter 3 and Part 3, Papers 2 and 3). Consequently, risk was defined as the likelihood that a specific level of effect will occur conditioned in the occurrence of a specific stressor level, in conjunction with the likelihood that this specific stressor level will occur (see Part 2, paper 2 and Part 3, 3.3).

So if E is the undesired effect and x is a level of stressor X, then the risk $R_X = L(E|x)*L(x)$, where L is a likelihood operator such as probability, possibility or necessity and * is a corresponding conjunction operator such as multiplication in the case of probability or maximum or minimum in the case of possibility and necessity.

R provides an estimate of the risk pertaining to that specific level of stressor ("instantaneous risk"). In order to assess the risk pertaining to a situation where a spectrum of stressor levels are possible, two approaches can be taken:

- The cumulative distribution of the instantaneous risk can be determined (this approach was used in Part 2, Chapter 3 and Part 3, Paper 2), or
- The maximum value of the instantaneous risk over all possible stressor levels can be determined, i.e. the likelihood that the system will experience the undesired effect can be no higher than the most likely instantaneous event. This is the basis of the fuzzy approach (Part 2, 3.4 and Part 3, Paper 3).

The Kelly-Roy-Harrison expression

Subsequent to submitting the papers in Part 3 the paper by Kelly and Roy-Harrison (1998) was discovered that gives a mathematical construct of ecological risk. This expression is meant to assess different consequences of a given stressor occurrence. If the consequences are discounted in one single end-point, it can be shown that this expression is a special case of the general inference scheme on which the above formulation is based (Part 2, Chapter3, 3.2)

4.3 END-POINT PROJECTION

One of strengths of the ecological risk approach is the requirement to establish clear end-points. This contributes to making the assessment transparent. As pointed out in Section 3.3 b) above, the statutory and surrogate end-points often do not coincide. An end-point projection model needs to set be up. An example of such a model is given in Part 2, Section 2.4.3 and Appendix 2, Sections 2.10.1 to 2.10.4 and Part 3, Paper1). This model is meant as a prototype to indicate what sort of inputs might be necessary and (qualitatively) how this might influence confidence in a risk assessment.

4.4 APPLYING RISK TO THE DIVERSE STRESSOR MULTIPLE SOURCE PROBLEM

A generalised scheme for the application of risk methodology in resource management and particularly with respect to establishing desired resource management stressor criteria, is shown in the figure below (see Part 2, Section 2.2.3)

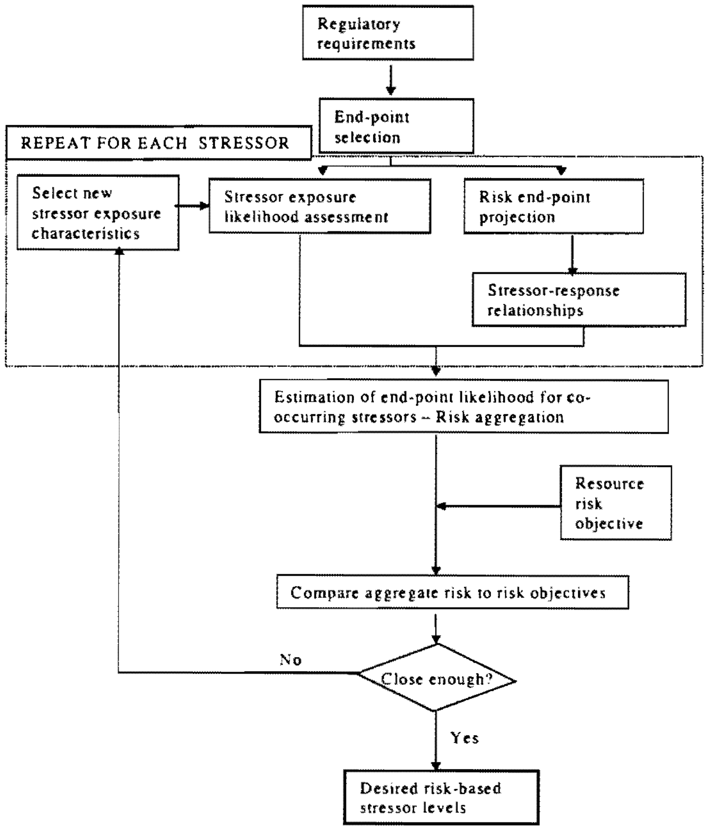

Diagram of a generic application of ecological risk-based management showing how aspects of the ERA process could be used. Detailed discussion appears in Part 2 Chapter 2.

4.5 AGGREGATE RISK

An important advantage in a likelihood expression of risk is the ability to compare stressors directly. The implication here is that identical end-points are used in the stressor specific risk assessment. Furthermore, stressor risk can be assumed to be logically independent, i.e. the occurrence of an effect due to one stressor does not imply the same effect due to any other stressor. (Logical dependence needs to be distinguished from mechanistic dependence where effects such as additivity, supra-additivity or infra-additivity might be at work and which will influence conditional effect dependence in the instantaneous risk assessment).

With this being the case, simple probability and possibility theory suggests modelling the aggregate risk as the disjunction (or union in set theoretical terms) of logically independent events. Examples are provided in Part 2 Section 2.5 and 2.6 and in Part 3, Papers 2 and 3 for probabilistic and fuzzy risk respectively.

4.6 APPLYING A RISK OBJECTIVE: THE DIVERSE-STRESSOR-MUTIPLE-SOURCE PROBLEM

Up to this point only a typical risk assessment scenario has been addressed where a situation exists where a stressor or stressors occur or may occur and the goal is to assess the resulting risk. However, the situation is somewhat more complex when one has to manage stressor levels to an ecological risk goal (Ecological risk-based management, ERBM).

This is analogous to waste-load allocation where an in-stream water quality objective is given and it is necessary to derive point source criteria to meet an in-stream objective. The problem now is that many different combinations of stressor-levels result in same risk. Therefore, additional information is required to decide on suitable source criteria. This apparent obstacle can be turned into advantage since it provides the opportunity to incorporate independent information (independent with respect to biological effect or exposure) into the assessment. Optimisation is required to solve this problem (see Part 2 Chapter 4 and Part 3 Paper 4).

The fuzzy optimisation problem was formulated as finding that set of stressor source attenuation values that maximised the overall acceptability of the regulated situation. It was assumed that the regulator would be satisfied when the risk was minimised but with a maximum threshold. On the other hand, the regulatees would be satisfied with minimised stressor attenuation with a graded acceptability between completely unacceptable and completely acceptable. Various ways of estimating the overall satisfaction were investigated, each relating to policy decision by the regulator.

Both Simplex and Genetic optimisation algorithms were explored but the genetic algorithm was found to be the most suitable.

5. GENERAL CONCLUSIONS

See also Part 2, Chapter 5 for more detailed discussion.

Is risk really conceptually useful in water resource management with the aim to ensure sustainability?

Ecological risk, formally defined as the likelihood that loss of sustainability will occur, is potentially very useful in the context of the NWA. In principle it addresses most of the major factors impacting on the uncertainty in ecological assessments at least semi-quantitatively. It could:

- Serve as a rational basis for classifying resources where the classification would take into consideration both what is known about the stressor effect on the system and what is known about the stressor's actual likelihood of occurrence.
- Be used in the management of highly utilised catchments as a tool to formulate policy and derive source and stressor specific management criteria.

Is there a mathematical construct that could be used for risk calculation in ecological risk assessment in the NWA context?

A theoretically sound way of assessing risk is presented in this study. It comprises a conjunctive stressor–specific risk estimation and a disjunctive risk-aggregation. This mathematical formulation is extended both to the probabilistic and possibilistic domains. It is computationally easy and it can be coded for spreadsheet use for resource classification purposes.

How could risk be applied in a multiple stressor multiple source environment?

a. **Ranking stressors** is simple enough on a risk basis.

b. Risk has the potential to be used as the basis for **stressor specific resource quality criteria**. The advantage would be that all stressors would then be comparable on the basis of the same effect. This aspect needs further development.

c. **Classification of resources** with a view to setting the reserve. In order to accomplish this it would be required to set ecological risk goals for resources and/ or classes of resources. This aspect needs further development.

d. Deriving source- and stressor-specific management criteria in catchments with high pressure for resource use. This would require co-operative effort from water users who have to be able to formulate ranges within which they are able to attenuate the stressors they produce. Computationally this is quite demanding but in cases where there is economic pressure this may pay off handsomely both to the regulator and the regulatees.

Two issue merit critical attention:

Deriving stressor-response relationships. Risk characterisation/ calculation remains critically dependent on the quality of the knowledge of the relationship between stressor occurrence and the corresponding response. In this study that knowledge was modelled either as a stressor-response relationship (that describes the likelihood of observing an end-point as a function of stressor level) or as a rule base formulating the same type of knowledge on a more qualitative basis. Methodology is needed to formalise the derivation of these relationships from experimental observation and/or expert opinion.

Deriving/ setting ecological risk objectives for streams. The success of risk-based management is critically dependent on acceptable risk objectives. Two aspects in particular need attention: acceptability to the water use community and acceptability to the scientific community.

Structure

The document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: (This Part) Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

Part 3: Presents some of the papers that have been published in peer reviewed literature and that are included for quick reference.

Part 2:

Technical discussion

CHAPTER 1

INTRODUCTION AND BACKGROUND

1.1 SUMMARY

In the South African context, the National Water Act supplies the regulatory background for water resource management. The provision of a suitable quantity and quality of water for basic human needs and sustainable use of the aquatic ecosystem as a Reserve, supplies the regulatory background for water resource management. This has to be balanced with the development needs within the water use community. The uncertainty and variability inherently part of the ecological knowledge base, which complicates this process, can be addressed by ecological risk expression. This supplies the basis for a continuous assessment of effect, which is necessary to find the optimal state between the satisfaction of ecological goals on the one hand, and the operational requirement for managing the system on the other hand. Specifically this study addresses:1) The systematic basis for deriving ecosystem level end-points from stressor occurrences, 2) Expressions of ecological effect likelihood and their convolution as a basis for the expression of overall effect expectation, 3) The optimisation procedure for estimating stressor attenuation levels in order to achieve ecological goals, and 4) An application framework for this derivation procedure.

1.2 INTRODUCTION

The South African National Water Act (Act 36 of 1998) (NWA) makes provision for the protection of a Reserve. The Reserve refers to a quantity and quality of water that will assure the supply of water for basic human needs as well as the sustainable functioning of the aquatic ecosystem (DWAF, 1997). The NWA contributes by giving effect to the right to a healthy environment as guaranteed by the South African Bill of Rights. In fact, the protection of the Reserve is the only right with regard to water under this Act. The NWA also does away with the *dominus fluminis* principle of the Roman Dutch law, which gives a riparian landowner the right to use of the water in the stream. Water is viewed as a resource to which all South Africans should have reasonable access and which is administered for the common good by the state.

1.3 REGULATORY BACKGROUND

In terms of the NWA, it should be noted that:

- ✓ The term "quality" is defined so as to include not only the chemical and physico-chemical components of the water, but also the integrity of biota, the assurance of flow and the habitat structure.
- ✓ The water resource includes, not only the water column of streams and rivers, but also the ground water, sediment and estuaries as well as the riparian habitat. Consequently, when reference is made to "resource quality", it encompasses virtually all manageable aspects of practically all compartments of the water environment (except the water/air interface).
- ✓ The aim of the NWA, besides the protection of the aquatic ecosystem and the supply of basic human needs, is to prevent or reduce pollution. "Pollution" refers to any alteration of the physical, chemical or biological properties of the resource that makes it harmful or potentially harmful to humans or aquatic organisms or the quality of the resource itself. The pollutants, or agents causing pollution by the definition above, are characterised by their ability to cause some form of stress (or adverse reaction) in the resource. The term "stressor" is therefore used further in the study as synonymous with "pollutant" strictly in the sense used in the NWA. This should be distinguished from a usage of the term pollutant, which mostly has the connotation of a substance that need only have a potential to cause harm.
- ✓ Under the NWA there is also a move toward a catchment management approach, as opposed to an exclusively pollutant source directed approach in water resource protection.

Although the concept of the Reserve makes provision for both human needs and that of the aquatic environment, the focus of this study is the sustainable function of the ecosystem and more specifically the application of risk methodology in water resource management. Most if not all the principles will be applicable to the human use part of the Reserve.

1.3.1 RESOURCE-DIRECTED MEASURES AND SOURCE-DIRECTED CONTROLS

The NWA makes provision for two sets of administrative tools to accomplish the goal of sustainable development of the water resource (DWAF, 1997):

1. Resource-directed measures (RDM's), which include a resource classification system that requires the grouping of significant surface water resources (among others) into protection classes. Each class represents a similar risk of damaging the resource beyond repair and corresponds to management objectives for water quality, quantity and assurance, habitat structure and biota. RDM's explicitly recognise that some damage has already occurred in the aquatic ecosystem (for example) but its point of departure is that no further degradation be allowed.

2. Source-directed controls (SDC's), which include source reduction measures that aim to reduce or eliminate the production of pollutants which could harm the water resource. SDC's will make use of permits and standards while promoting changes in technology and land-use.

> Resource-directed measures in the context of the ecological aspect of the Reserve would focus on resource protection and supply the basis of instream management objectives. The source-directed controls supply the executive means of realising resource protection. Quality criteria would necessarily be an integral part of both resource-directed measures and source-directed controls.

1.3.2 REGULATORY IMPACT OF THE RESERVE

Section 15 of the NWA makes it mandatory that any action that follows from the Act must give effect to the RDM class and its associated water resource quality objectives while Section 18 demands that such actions must also give effect to the Reserve. Section 16 determines that the Reserve must also be set in accordance with the class.

In making regulations on water use, besides giving effect to the Reserve and the resource classification system, Section 26 requires that, inter alia, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the instream and riparian habitat. Water use regulation must take into account factors such as (Section 27. (1)):

1. The socio-economic impact of water use or curtailment of use (d)
2. The catchment management strategy applicable to the resource (e)
3. The likely effect of the water use on the resource and other users (f)
4. The class and resource quality objectives (g)
5. The investment already made and to be made by the water user (h)
6. The quality needs of the Reserve and to meet international obligations (j)

> The regulatory requirement is that the SDC's must give effect to the RDM's but both of these must give due consideration to their impacts on the ecosystem and the water users. While SDC's have to give effect to the RDM's, they could be wider in their reach than RDM's and could take into consideration technology issues.

1.3.3 THE "DEVELOPMENT VS. PROTECTION" DILEMMA

From the foregoing and an analysis of the provisions in the NWA (See Appendix to Chapter 1) it is clear that:

⇨ The Reserve is central to water resource management in South Africa. The Reserve is the quantity and quality of water necessary to provide for basic human needs and the protection of aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource. The reserve must be given effect, not only on a site-specific basis, but also at catchment level.

⇨ The aspects of water that needs to be managed are diverse, including flow-, substance-, habitat- and biodiversity-related stressors. These stressors have to be managed in a way that ensures sustainability.

⇨ The use of the term "sustainability" implies that pressure on the ecosystem is expected and allowed. Moreover, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the insert and riparian habitat. Water use regulation must take into account factors such as the socio-economic impact of water use or curtailment of use, the likely effect of the water use on the resource and other users, the class and resource quality objectives and the investment already made and to be made by the water user.

It is intuitively clear that resource protection, as typified by the Reserve, may somehow have to be traded off against resource development in support of other development needs. This is by no means a new problem. A simplistic formulation of this problem is "protection" (represented by a set of standards or criteria, usually with reference to the chemical and physical characteristics of water), versus "development" (represented by some economic or social surrogate measures such as "treatment cost" or "jobs lost").

Broadly, the RDM's represent the protection requirement. The SDC's on the other hand have to deal with the reality of setting end-of-pipe criteria among others, which are important for the design and operation of effluent treatment plants, for example. These relate to the economic and technical issues, which finally have socio-economic impacts. The NWA requires that RDM's and SDC's be coherent. However, in keeping with its approach to all technical matters, the NWA does not prescribe the possible approach needed to solve the problem of aligning the Reserve, RDM's and its corresponding resource quality objectives with the SDC's (such as waste discharge regulations) needed for the practical enforcement of the law.

At present the management objectives corresponding to the ecological RDM classes are set in terms of the South African Water Quality Guidelines (SAWQG, 1996; MacKay, 1999). The use of these substance/ stressor specific guideline criteria must be seen against the background of two issues: 1) The management context and 2) The diverse-stressor-multiple-source problem.

1.4 MANAGEMENT CONTEXT

Two aspects of the management in the context of the ecological Reserve are described: 1) The factors impacting on objectives and criteria in resource management and 2) Basis for formulating objectives and criteria.

1.4.1 FACTORS IMPACTING ON OBJECTIVES AND CRITERIA

The goals set by the NWA need to be translated into objectives. The objectives are the achievable "milestones" in attaining the goal. The objectives need to be translated into criteria, which are practical management values giving effect to the objectives.

The NWA goal "protection of ecological sustainability" might, with a number of assumptions, be translated to the objective "protect 95% of the aquatic species most of the time". This objective would give rise to the criteria as given in SAWQG (1996).

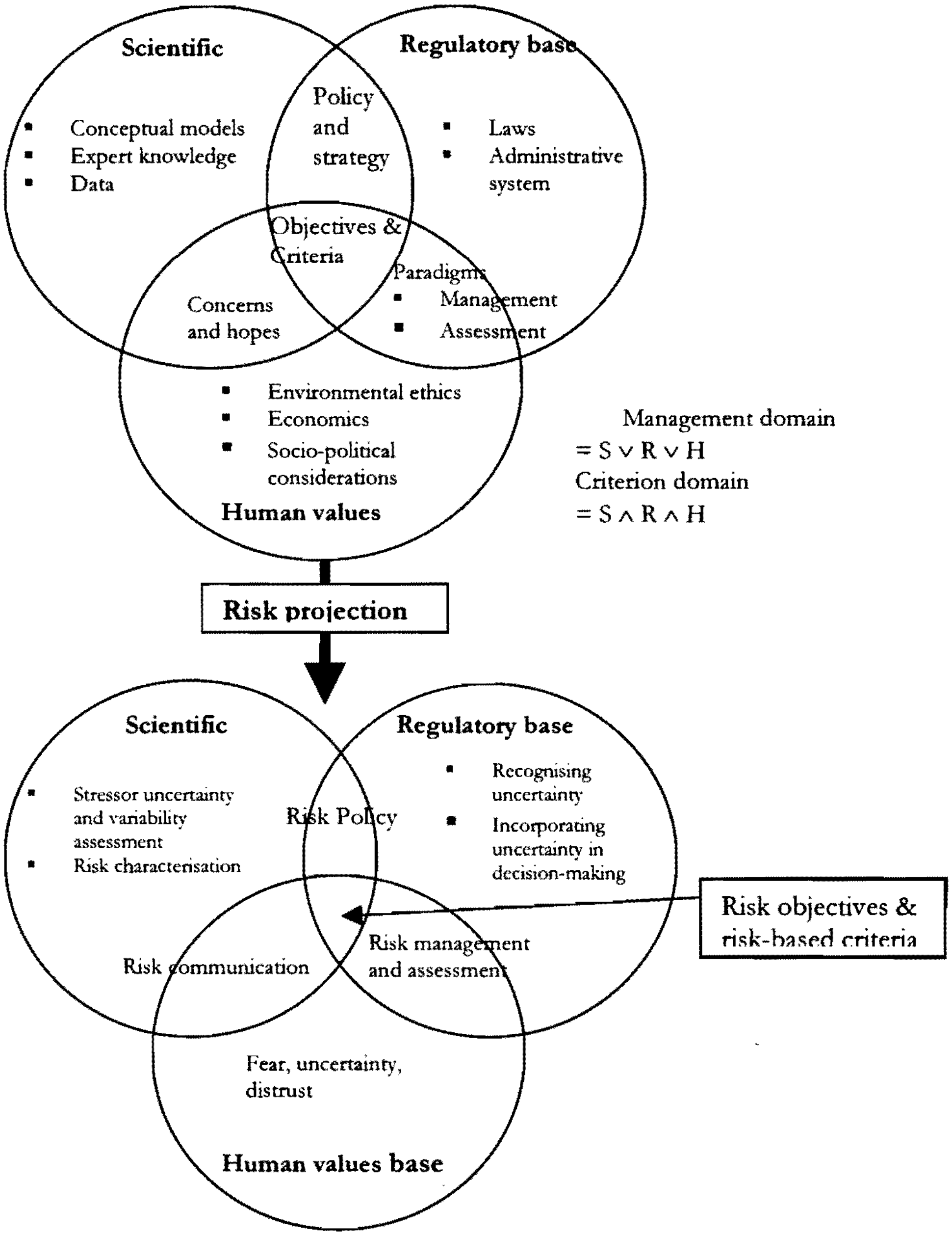

Figure 1.1 Some input domains of water resource management and how they relate to the application of risk-based decision-making

A conceptual model of the basis of management criteria is shown in Figure 1.1. The resource management domain is depicted as the conjunction of three of separate bases or domains, the boundaries of which are naturally fluid and fuzzy:

1. The **scientific base** which deals with the gathering and systematising of ecological and other environmentally significant knowledge. This area will include most of the fundamental sciences like chemistry, physics, biology, geology and mathematics as well as some of the applied sciences like environmental chemistry, toxicology, hydrology, hydraulics, statistics, information technology, soil chemistry and physics, geomorphology, limnology and the like. These would be the group sometimes referred as the "hard" sciences.

2. The **regulatory base**, which deals with the laws and administrative systems, put in place both ranging from laws promulgated at central government level, down to operational rules of companies. These supply the infrastructure within which the day-to-day running of society takes place. It is likely that disciplines of macroeconomics, state administration and international affairs and political science would have an impact at this level.

3. The **human values base,** which deals with the way individuals and communities organise their lives and the way in which they view and would wish to manipulate their environment. Disciplines such as ethics (particularly environmental ethics), microeconomics and probably socio-political considerations would have an impact at this level. These are sometimes referred to as the "soft sciences".

Objectives and Criteria for resource management are impacted by all three domains and particularly by the interfaces between domains.

Policy and strategy is used here in the sense of technical policy and management strategy. These determine how some areas of uncertainty are to be handled in terms of, for example, assumptions that need to be made (e.g. when insufficient data are available, then a precautionary approach might be used or, to curb eutrophication, the use of phosphate builder in soaps might be phased out). The use of resource directed measures and source directed controls in water resource management are also a matter of management strategy.

The **management and assessment paradigms** stem largely from the way the human values interact with regulatory system, but it may (and should) be influenced by scientific knowledge. The assumption of a blanket precautionary approach, for example, may be influenced by a) a knowledge that the economy of the country as well as the socio-political situation will allow it, b) human environmental ethics dictate that "only the best is good enough for the environment" and in conjunction with this c) the legal system and regulatory framework require minimising possibly conflicting technical/scientific input. Furthermore, it might be required that an environmental assessment yield a clear acceptable/unacceptable answer because of the human mind's conditioning to see clear and unequivocal answers as the only expressions of certainty particularly in legal/ litigatory situations.

On the other hand, the interface between human domain and the scientific domain determines the **fears and hopes** both of the "lay" public and the "experts" who are, of course also human. This interfacial area also typically contains the area of science philosophy, which has an impact both on what is considered "good" science and what is considered "relevant" science.

> A criterion is a crucial component in regulatory administration that may have far reaching effects for the regulatee. While regulatory and scientific inputs may dominate in many cases, the derivation of viable criteria needs to recognise the importance of human values input. Practicable criterion derivation methodology should ensure that input from the human sciences can be accommodated in what might otherwise be a highly technical process.

1.4.2 BASIS FOR FORMULATING MANAGEMENT OBJECTIVES AND CRITERIA

Decisions and hence the formulation of the associated objectives and criteria in the management of the water resource could be:

1. **Bureaucracy driven**: i.e. management process is driven by the need for its own existence and is largely an administrative process. The bureaucracy driven approach is not a functional approach and when it does occur, it is more likely to be an artefact of a degraded administrative process and does not merit further discussion.

2. **Technology driven** i.e. the available technology and economics of the technology dominates decision-making while the effect of stressors on the system, is accommodated to the extent possible. The way in which effluent management criteria are set will therefore mirror the decision-making approach. Various technologies may be prescribed for emission impact reduction at source, such as Best Available Technology (BAT), Best Practical Means (BPM), Best Available Technology Not Entailing Excessive Cost (BATNEEC), as well as a number of other qualifying variants of the above (Foran and Fink, 1993). Presumably, the rationale in using technology-oriented decisionmaking (and effluent criteria) is that if the technology does not exist to effect a management action, that action is simply not viable. The disadvantage of such a

technology approach is that it does not necessarily achieve management goals and does not in itself supply the need for technology development.

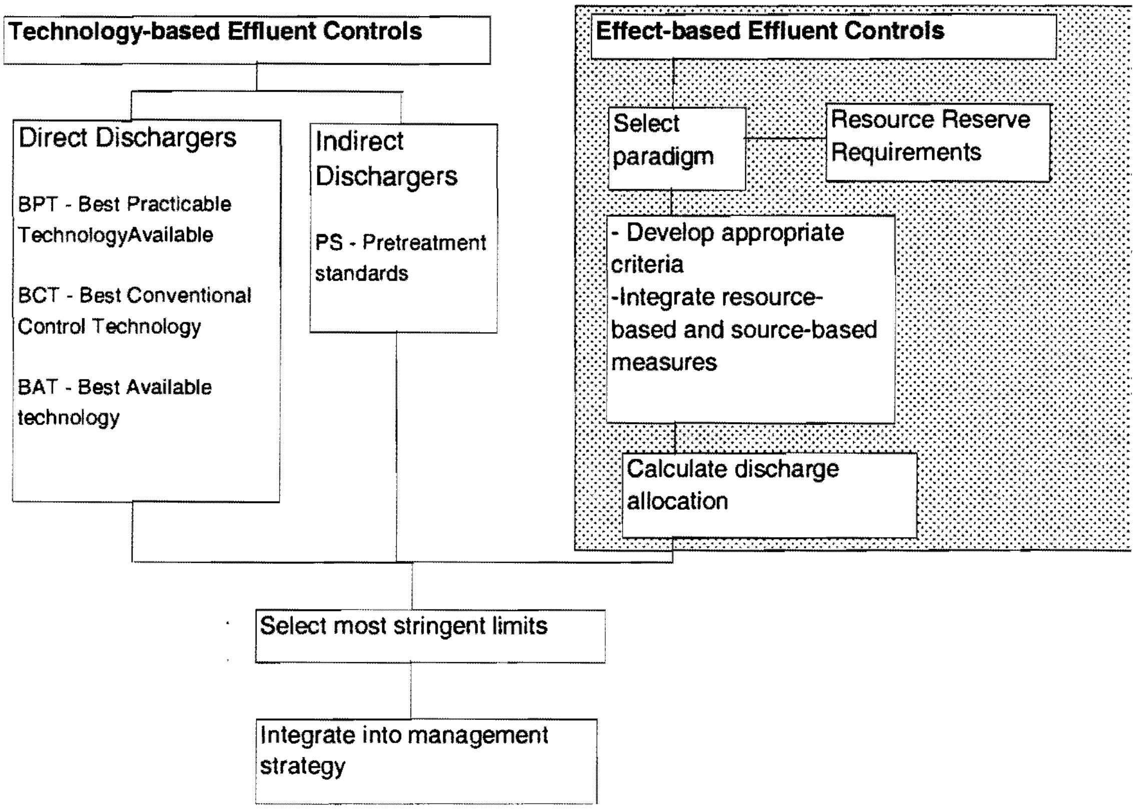

Figure 1.2. A diagrammatic of approaches to effluent management (adapted from Foran and Fink, 1993). The focus of this work concentrates on the shaded area. SDC's would be involved in the final step and could therefore draw on the output of this study.

3. **Resource driven** i.e. some valued function or process of the resource such as water use or economic activity rather than available technology drives management decisions. The effect of a stressor on the system dominates decision-making while technological limitations are recognised. Effect-driven decision-making (and effluent criteria) usually considers what the requirement is in-stream for some defined use of the water. This requires that some environmental quality objectives (EQO's) are set (Strortelder and Van der Guchte, 1995; Ragas, *et al.*, 1997). The EQO approach has been used in the UK while the technology based approach has predominated in countries such as Austria, Belgium, Germany and the Netherlands. In the USA, both approaches have been used in parallel (Foran and Fink, 1993). Technology based criteria are set and then the likelihood of violating EQO's are assessed. If the EQO's are likely to be violated

then the EQO approach is used to set criteria, otherwise the technology-based criteria are used. The latter two approaches are contrasted in Figure 1.2.

From the point of view of the resource management to achieve the Reserve goal, it would be preferable to follow an effect-based (e.g. environmental quality objectives or EQO) approach rather than a technology based approach. This has been suggested for use in South Africa (Van der Merwe and Grobler, 1990). The goal of the NWA is to achieve a specific effect, i.e. to maintain sustainability in the ecosystem. Consequently, the EQO approach has to be adapted to the characteristics of the ecosystem and ecological processes, as well as the needs of the catchment, particularly:

- It needs to recognise that not only the chemical and physico-chemical composition of water is involved, but that a diverse range of stressors might be involved,
- There is a natural variability in environmental conditions (including a specific frequency of extreme events such as floods and droughts), that is not only innocuous but necessary (CSIR, 1989).
- While resource objective driven decisionmaking may supply the impetus for technology development, it is still dependent on the technology necessary to achieve these goals. This implies that a purely effect-driven approach to setting EQO's may not be viable. The limitations and implications of underpinning technology need to be recognised.

1.5 THE DIVERSE-STRESSOR-MULTIPLE-SOURCE (DSMS) PROBLEM

While stressor-specific point-source criteria or standards are administratively advantageous, it can be shown (Part 2: Paper 1) that it is no guarantee of desired in-stream effect. For this reason, the concept of in-stream water quality objectives was used. The in-stream objective could be set to correspond to the level of a water quality variable which is expected to provide the desired level of protection (with perhaps a safety factor added). Establishing the end-of-pipe criteria corresponding to these objectives necessitates the use of waste load allocations (WLA's). The total load corresponding to the objective concentration (in the case of stressors in solution) can then be apportioned among the sources of such stressors. However, in terms of the Reserve required under the NWA, the conventional WLA to stressor specific water quality objectives is at a disadvantage because of:

- The **additivity effect** of a number of **similar stressors**. E.g. the combined effect of a number of different toxic substances which are discharged to a river (each of which complies to its own particular acceptable effect concentration) may be greater than acceptable due to some form of additive or supra-additive (or even synergistic) effect. This problem on its own is not insurmountable since stream objectives may be adjusted to accommodate this phenomenon but it becomes administratively cumbersome.

- The **diverse-stressor (DS) problem**. Even when additive effects among toxicologically similar stressors are accounted for, estimating the combined **effect of dissimilar stressors** may be impossible. The action of the stressors may be mechanistically dissimilar although the final effect may be the same. A WLA in itself cannot overcome this problem.
- The **diverse-stressor-multiple-source (DSMS) problem**. When a number of heterogeneous stressor sources have to be accommodated, this exacerbates the DS problem. Now a common basis for expressing impacts is called for in order to optimise the apportionment of stressor attenuation. Stressor metrics (such as concentration and flow) is no intrinsic **common basis for comparison** on which WLA may be based. When apportioning toxic substance load, nutrient load and flow deficiency (all of which may result in ecosystem stress), for example, the stressors are dissimilar both in units of measurement and mechanistically. Not only is the effect of diverse stressors not accounted for, but the allocation of the stressor load among different sources can lead to an infinite number of combinations of stressors that are all equally valid.

Fundamentally, the problem described here is that the WLA tends to be dominated by the stressor rather than by its effect. Changing from an stressor- to an effect-oriented approach may solve the problem since a fundamental rationale of water resource management (or any other resource management for that matter) is to achieve a specific goal by managing the inputs.

1.6 RATIONALE FOR THE USE OF RISK METHODOLOGY

The rationale for using risk-oriented methodology is argued in Part 3, Paper 1. Some of the main points are listed here.

1.6.1 A RISK APPROACH

A risk approach is used here as a counterpoint to a hazard approach to resource management. A hazard in this context refers to the potential that a stressor has to cause some unacceptable effect. The SAWQG criteria are examples of hazard-based criteria.

HAZARDS AND HAZARD-BASED CRITERIA

The criterion derivation process for the SAWQG's used toxicity data, but by assumption specific benchmarks of effect (such as LC50 values in the case of the Acute Effect Value or AEV) were selected as the basis for criterion derivation. The AEV would be an indication of maximally acceptable hazard. All the uncertainty relating to the data and derivation process has been discounted by precautionary assumptions (Roux, *et al.*, 1996).

By definition any single hazard-based criterion recognises only one type and level of effect (e.g. mortality at the 50th percentile in the case of the AEV). Consequently only the stressor and its characteristics are considered variable. A hazard-based criterion would therefore typically be a stressor value corresponding to a level of acceptable effect (e.g. the general AEV for cadmium in moderately hard water is 6µg/l). There is no indication of how the hazard changes as the stressor value changes, for example. The hazard either exists or it doesn't. So, when apportioning the load, using a hazard criterion gives no indication how disastrous it would be if the objective were temporarily exceeded by 10%, or 20% or even 50% This would normally call for expert opinion and it is a soluble problem, but the solution is not implicit in the problem formulation

If the assumptions in the derivation process are explicitly precautionary, then the criteria are useful in setting the most stringent on a stressor-by-stressor basis. As such, they may define the most conservative end of the management objective spectrum.

> Hazard-based criteria are useful management tools inasmuch as they may represent the precautionary objectives for resource management. However, they may lack the flexibility necessary for the management of diverse stressors in a multiple source environment.

The type of criterion is also closely associated with the paradigm in which it is used (See the quantal assessment paradigm (QAP) and the continuous assessment paradigm (CAP) described in Appendix A1.2). Hazard-based criteria are necessarily associated with the QAP (although the use of the QAP does not necessarily imply the use of hazard criteria). While it is useful to have fixed values of variables to assess situations for law-enforcement, it must be recognised that this does not make the best use of all the available scientific information.

RATIONALE FOR THE RISK APPROACH

In characterising the Reserve and managing for its sustainable use, some fundamental characteristics of the ecosystems and ecological assessments need to be noted:

1. There is an innate and practically irreducible inter- and intraspecific variability in biotic response to a given stressor as well as in many other aspects of in biotic systems (O'Niell *et al*, 1979;

Kooijman, 1987, Levine, 1989; Brown, 1993). (These concepts are discussed more extensively in Chapter 2.)

2. In many natural ecosystems there is a dearth of detailed data about structure, function and composition that adds to the overall uncertainty regarding ecosystem models and their predictions, which limits the scientific certainty about any biotic system and its responses.
3. The response of organisms to stressors is normally continuous and discontinuities are normally an artefact of the scale or means of observation (notwithstanding the possibility of a threshold of effect). Generally, there are no natural discretisations in the continuum of response..

The consequence of this is that a deterministic, quantal view of management actions and their consequences may be inappropriate. A more probabilistic, continuous approach as typified in the continuous assessment paradigm (CAP, see Appendix A1.2.1) is indicated. Risk is a suitable basis for ecological assessment in the context of the Reserve and RDM's since it:

- Is by definition, is a probabilistic expression and therefore caters uncertainty and variability explicitly (See e.g. Bain and Engelhard, 1987).
- Allows for a CAP (Suter, 1993) since it allows the use of all the stressor response data as well as the exposure data.
- Is explicitly effect-based as it requires an explicit end-point, which could incorporate the human concerns.
- Probability theory allows for events (such as the occurrence of a selected end-point dependent on the occurrence of different stressors) to be partitioned into component events (such as the occurrence of the end point dependent on single stressors or selected groups of stressors). A theoretical underpinning exists for establishing the relationship between the main event and the component events (see Chapter2).

> It is postulated that risk as a more suitable basis on which to base objectives and criteria related to resource management compared to hazard, since the characteristics of risk is better suited to the ecological assessment domain than hazard. This supposes that risk objectives analogous to hazard objectives can or have been set.

1.7 GOAL AND OBJECTIVES

The aim of this study is to introduce, at a conceptual level, the use of risk or risk-related methodology to solve the DSMS problem (in 1.5 above) in the context of the ecological Reserve required under the South African NWA or in any situation where risk objectives can or has been set for a water resource.

In particular, source-specific criteria are envisaged that correspond to ecological risk objectives set for the water resource, while at the same time recognising that technological or other factors may determine the level of acceptable stressor reduction.

These source management criteria are not meant to supplant any other resource criteria (such as the SAWQG criteria for the protection of the aquatic ecosystem). Such water quality objectives may still form the basis source-specific waste load allocation of individual stressors where appropriate. The risk-based source-specific criteria will likely only be applied in a catchment management context and only when: a) there are indications that several diverse stressors may all contribute to an impact on the water resource, or b) there is conflict among source managers and regulatory authorities.

1.7.1 GOAL

The problem to be solved can therefore be formulated as: **Find a rational means to derive stressor-source management criteria that give effect to the Reserve concept in a catchment when there are multiple (diverse) stressors originating from a number of identifiable and manageable sources present in a catchment, taking into account that management criteria have definite socio-economic as well as technical implications.**

1.7.2 OBJECTIVES

In order to achieve this goal, the following objectives need to be met:

- The formulation of end-point projection problem. How to relate the likelihood of effect at a higher ecological level when only data for the estimation of a lower end-point is available (Chapter 2).
- Formulating stressor-response relationships. The estimation of the likelihood of effect is a fundamental requirement of the ecological risk (Chapter 2).
- Solving the diverse stressor problem. How to estimate likelihood of a specific effect when diverse stressors occur together. This amounts to a mathematical formulation of the ecological risk characterisation step in the ERA process (Chapter 3).
- Formulating DSMS problem as an optimisation problem and solving the optimisation problem (Chapter 4).

CHAPTER 2

BACKGROUND AND THEORETICAL CONSIDERATIONS

If a man will begin with certainties he shall end in doubts; but if he will be content to begin with doubts he shall end in certainties – SIR FRANCIS BACON

2.1 SUMMARY

In this chapter the difference between ecological risk assessment (ERA) and ecological risk based management (ERBM) is investigated further. The effect assessment phase would include formulating a stressor-response relationship (SRR).

Two major issues in formulating the SRR are: a) deriving a relationship between the likelihood of observing an end-point at higher (both conceptual and organisational) levels when only lower level data are available, and b) informing the SRR's.

The end-point projection problem is formulated in both probabilistic and possibilistic frameworks. The obvious point is demonstrated that the confidence in the risk with higher-level end-point cannot be greater than the risk predicted from lower level data.

Data for informing toxic SRR's will need to be derived from toxicity bioassessment, but careful attention needs to be given to factors such as level of organisation of the end-point and time variable toxicity levels.

Flow and habitat SRR's are likely to depend on expert opinion. It is therefore necessary to establish methodology by which to update the SRR's from field observations. Dempster-Schafer and other updating methods may be applicable.

2.2 INTRODUCTION

2.2.1 ECOLOGICAL RISK ASSESSMENT VS. ECOLOGICAL RISK-BASED MANAGEMENT

ECOLOGICAL RISK ASSESSMENT

Risk assessment is a well-established tool in both economics and engineering. The application of risk assessment to ecological assessment, ecological risk assessment (ERA), is a tool in environmental management. It is mostly used in the context of predictive risk assessment when a stressor is given. The framework and techniques of ERA have been widely used and are well known (Suter, 1993; Crouch, *et al.*, 1995; EPA, 1996; EPA, 1998). A simplified process diagram for ERA appears in Figure 2.1 while Figure 2.2 adds some more detail to show the interrelationship between ERA and risk management.

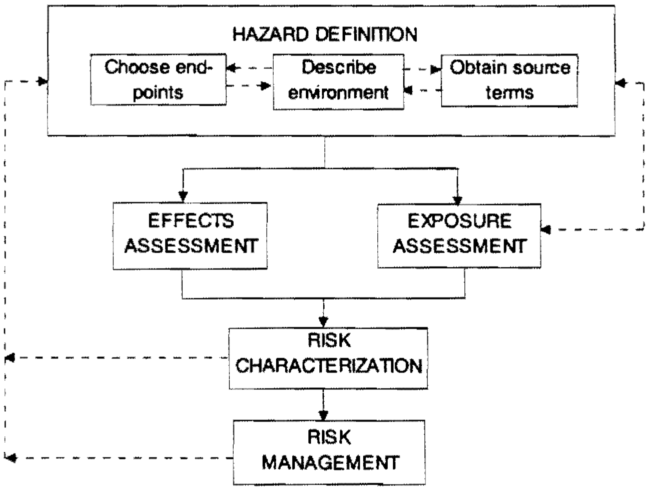

Figure 2.1. A simplified diagrammatic representation of the process of ecological risk assessment illustrating the main steps. The dashed arrows indicate feedback loops in the risk assessment paradigm. (From Suter, 1993).

ERA provides a structured methodology to formulate the societal values in measurable end-points and then to assess the likelihood of the occurrence of this end-point (EPA, 1998). The expression of risk in terms of likelihood stems explicitly from recognising the impact of uncertainty and variability (see 2.3 below) on the outcome of the assessment. This stands in contrast to some forms of environmental impact assessment that takes great pains to enumerate the potential impacts, but stops short of making an explicit assessment of the impact of uncertainty and variability on the overall situation assessment (DEAT, 1992; DEAT, 1998).

ERA has been used extensively in the management of stressors (pollutants) in the environment. It supplies a relatively objective means to compare different stressors, sources or treatment techniques. The methodology incorporates the best available knowledge on the source, environmental partitioning, and ecotoxicology of a stressor, the ecology of the receiving environment as well as societal concerns and issues and expresses it as a risk.

The expression of risk as used commonly in ERA involves some concept of likelihood of an effect on a target entity in the ecosystem, while the dimension of the stressor does not necessarily have to appear. For example the result of an ERA might be: "The probability of the loss of 10% of species due to stressor A is 0.01 while the probability for the same end-point due to stressor B is 0.2". In this way, it supplies a common basis for the comparison of otherwise dimensionally incompatible stressors. At the same time it is also a basis for communication of a rather technical process with a (possibly) technically illiterate or semi-literate audience.

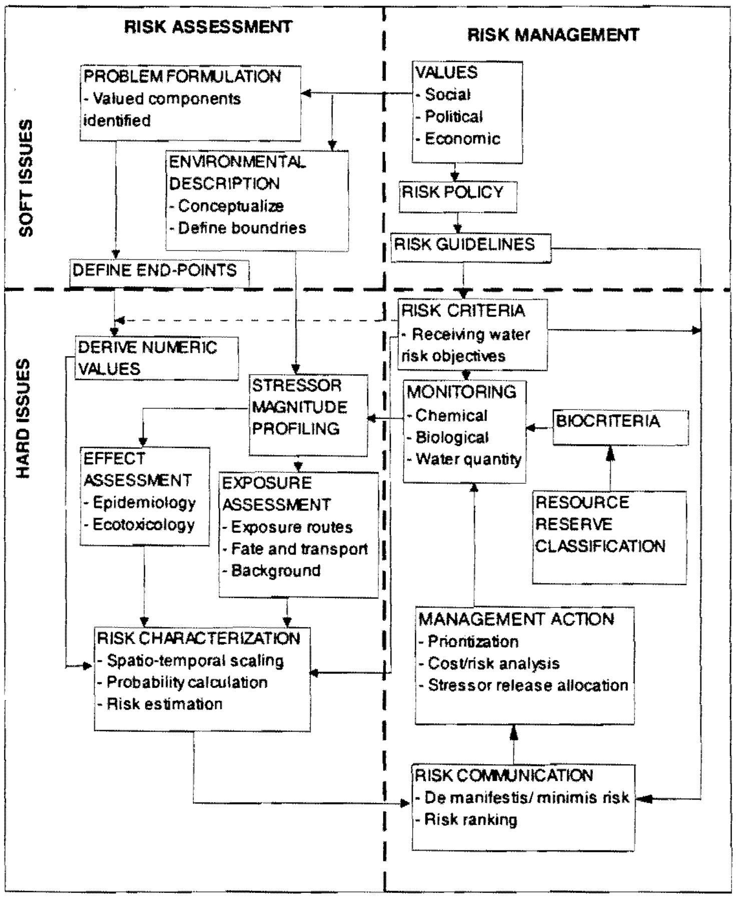

Figure 2.2 A more detailed analysis of the hard and soft issues involved in predictive ERA and its relation to risk management.

The rationale for applying ERA stems from the implicit question: "If stressor X occurs and effect E is the allowable effect, what is the likelihood (perhaps expressed as probability) that X will result in E?" In this case the stressor will be characterised by measured or predicted values of X.

2.2.2 NOTES ON CONVENTIONAL ERA

The main features the ERA process (Figure 2.1) include:

1. The **hazard definition** or **(problem formulation)** phase where an **end-point** for the assessment is selected, the **environment** in which the assessment is performed is described and, in general, the **stressor source** is characterised. The end-point includes both a **target ecological entity** and a **specific effect**.
2. The **effect assessment** phase in which (among other things) the relationship between the magnitude of the stressor and the likelihood of observing the end-point is identified.
3. The **exposure assessment** phase, where the likelihood of exposure of the target entity to the stressor is characterised.
4. In the **risk characterisation** phase the effect and exposure data is convoluted to obtain a quantitative or qualitative risk estimate (among other things).
5. The risk estimate is fed back to the **risk management** phase where the risk assessment request most likely had its origin.

With regard to the **hazard definition** or problem **formulation phase** it is noted that:

(a) An assessment end-point is required which, whatever that target entity is, has unquestionable or at least consensus value within the decision-making group (the upper right quadrant in Figure 2.2).

(b) Explicit provision is made for ecological models in the problem formulation phase of ERA that ensures that all routes of exposure to all relevant ecological compartments are addressed (Suter, 1996).

(c) Conceptual model development, which consists of formulating and contextualising the risk hypotheses. Risk hypotheses (*inter alia*) are assumptions about the consequences of risk assessment end-points and may be based on theoretical models, logic, empirical data or probability models. In complex systems, they are likely to be strongly dependent on expert judgement. The point of these hypotheses is ultimately to structure the analysis. It provides a link between the actual knowledge and problem it sets out to solve. In addition, they are useful in accounting for and characterising the uncertainty in an assessment.

With regard to the **effect assessment phase** it should be noted that:

i. All the available data should be used to establish the relationship between the selected end-point and the stressor occurrence

ii. All lines of evidence should be investigated. This might include information from laboratory studies, direct field observation of stressor-target entity interactions at the risk assessment site or inferred interaction from other suitable sites.

iii. All of the above can in principle be synthesised into a stressor response relationship (SRR), which is an expression of the functional relationship between the level of a stressor and the expected impact on the end-point effect on the target ecological entity. This might, for example, be expressed as a mathematical function or a rule base.

With regard to the **risk characterisation** phase (Suter, 1995):

i. The simplest form of expressing risk is by a point estimate such as the ratio between the expected stressor level (ESL) and some benchmark effect level (BEL). In this form it takes no cognisance of the uncertainties in variability involved in the assessment.

ii. Taking uncertainty into consideration, risk could be expressed as Likelihood (ESL > BEL).

iii. There does not appear to be a formal, generally accepted formulation of the relationship between risk, the SSR and the stressor exposure distribution.

2.2.3 RISK-BASED MANAGEMENT UNDER THE NWA

If risk is to be used to harmonise RDM class goals with SDC criteria, then it is implicit that a risk should be given as a goal. The RDM classification protocol contains the sense of risk implicitly. The basis for classification is the risk of destroying the Reserve. This risk is here defined as the **resource class risk objective**.

In the process of establishing the relationship between RDM's and SDC's it is necessary to establish the characteristics of the stressor given a risk objective. This process will be referred to as ecological risk based management (ERBM). Here the implicit question is somewhat different: "If effect E with likelihood R is all that can be allowed, what should the characteristics of stressor X (perhaps expressed as probability) be to accomplish this?" The ERBM process is very similar to the ERA process (Figure 2.1) except that the risk characterisation step and the flow of information is essentially the reverse of that for ERA (Figure 2.3).

When several stressors occur together in a water resource for example, available methodologies allow for a risk assessment for each individual stressor to be performed. It appears to be feasible to make use of the likelihood expression of risk to obtain an indication of the likelihood of the end-point phenomenon. With a management goal oriented choice of end-point, the integrated risk with respect to this end-point may then be a rational basis for apportioning the use of the water resource.

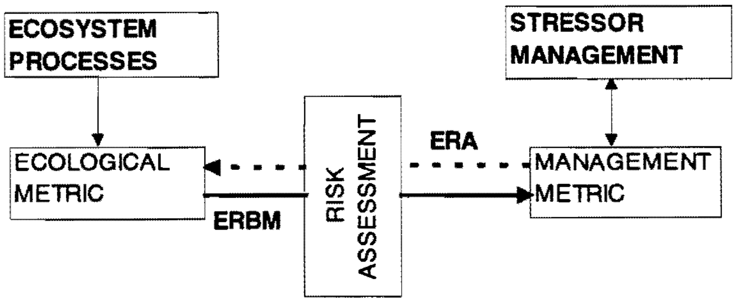

Figure 2.3 A comparison between the ecological risk assessment flow of information (dashed lines) and that of ecological risk based management. Some form of risk assessment framework remains the interface between the management metric (such as stressor release rate) and the ecological metric (such as sustainability or resilience). The risk assessment interface for ERBM is expanded in Figure 2.4

In its most fundamental form, a risk numeric value is calculated from some form of convolution of an effect likelihood expression (e.g. a probability distribution) and a stressor occurrence likelihood expression. If risk is expressed probabilistically, then deconvolution for the ERBM process could be very difficult. It would involve calculating every combination of effect probability-stressor probability that could result in a particular risk probability.

It can be concluded that:

(From Chapter 1) in the application of risk methodology under the NWA both the target ecological entity and the end-point is fixed. The target ecological entity is the ecosystem and the end-point is sustainability

The approach in ecological risk-based management (ERBM) is in a sense the converse of ERA. The point of risk-based management is to assess the level of stressor corresponding to an accepted level of risk.

In both ERA and ERBM stressor response relationships (SRR's) are important. A formalised structure for relating the regulatory end-point to the experimental/ observation level end-point. The ways in which the SRR is informed from observational data needs to be considered.

For ERBM under the NWA it is necessary to be able to express the aggregate risk. A mathematical expression of aggregating individual stressor risk is needed.

An expansion of a generic ERBM process might be summarised as shown in Figure 2.4. This study concerns itself with the shaded areas in this diagram.

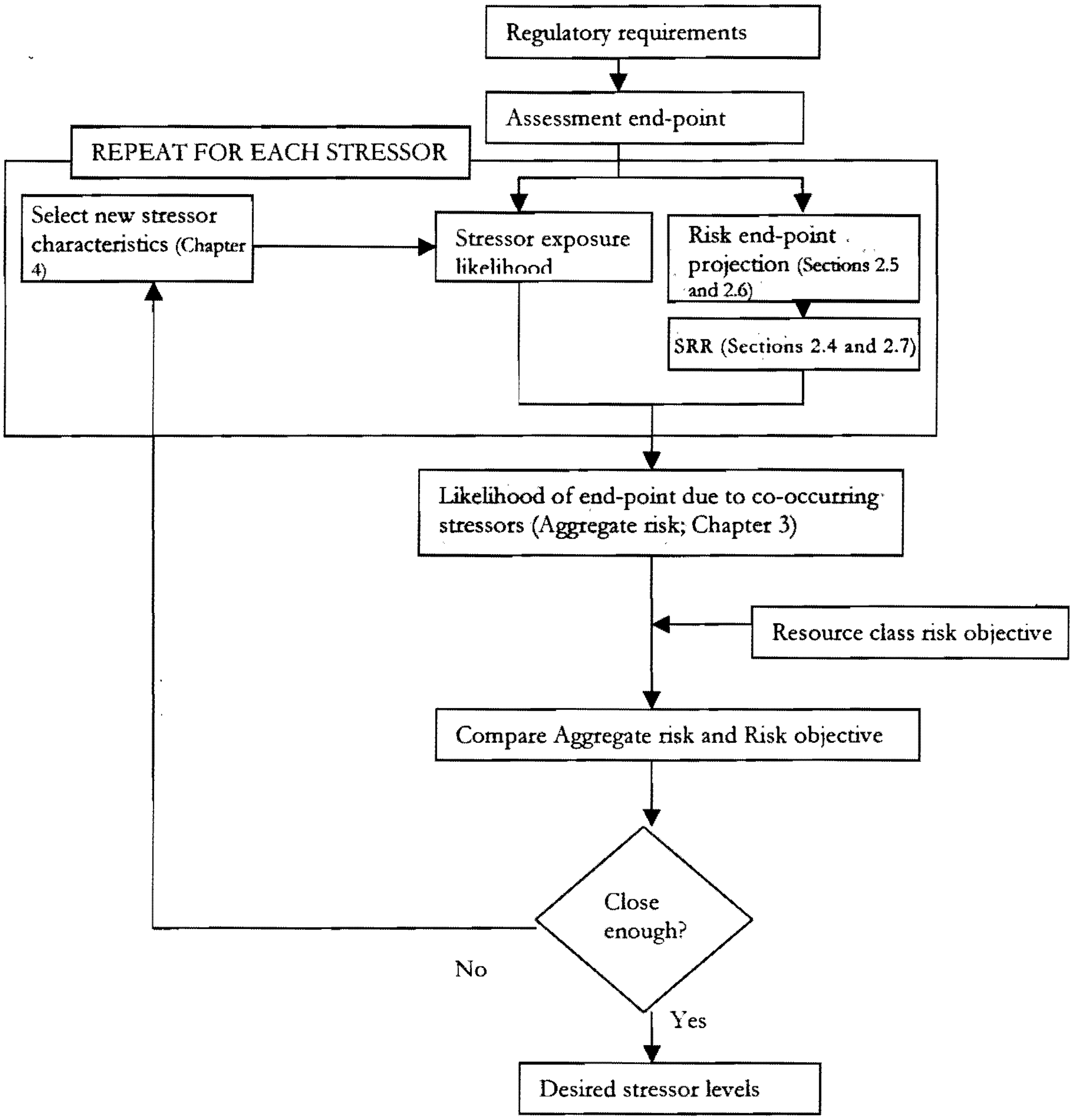

Figure 2.4. Diagram of a generic application of ERBM framework showing how aspects of the ERA process are used. This study concerns itself with the shaded areas in the diagram.

2.2.4 RISK AS LIKELIHOOD

Although many of the formal definitions of risk (such as those referenced under definitions) emphasises the probability aspect of risk assessment, the general problem is in estimating likelihood of adverse effects (Suter, 1995). The term "probability" has come to be associated in technical literature with precise but stochastically distributed observations. In the management of ecosystems this definition cannot always be met (See Chapter 1). System specific knowledge may at times be imprecise or uncertain and not necessarily influenced by randomness. In view of the discussion in 2.3 below, it is fitting that the term "likelihood" rather than "probability" is used in referring to ERA in general.

2.3 UNCERTAINTY AND VARIABILITY

It has been recognised that the rationale for risk assessment stems from the need to incorporate the effect of uncertainty and variability on decision-making (Frey, 1993; CRARM, 1997; EPA, 1998).

Colloquially, variability may be seen as a source of uncertainty in an estimation. Within the risk assessment community there is a distinction drawn between uncertainty and variability (Frey, 1993).

The phenomena referred to in the conceptual Reserve-related end-point formulation may be subject to either or both uncertainty and variability. With reference to ecological risk assessment, it has been recommended that uncertainty and variability be separated to provide greater accountability and transparency in a probabilistic assessment (USEPA, 1997b).

2.3.1 VARIABILITY

Variability is recognised as a natural characteristic of biota (e.g. Brown, 1993, Grimm and Uchmanski, 1994, Kooijman, 1994). Several forms of variability could be encountered. There is variability in the individual response of the biota to a given stressor exposure (Hathway, 1984) which is evident in the classic dose response curve of toxicology. Other stressor-response curves may, in principle, appear similar although the curves need not necessarily be strictly monotonic. Although these functions may not necessarily be measurable in controlled laboratory experiments, a combination of field observation and expert interpretation is likely to provide an estimate of the stressor response relationships. In this regard the use of a Bayesian statistical approach rather than a strict frequentist approach may be indicated (Frey, 1993).

Variability has the following characteristics:

- It is **inherent characteristic** of the system being observed.
- It stems from an underlying **stochastic mechanism** in which the outcome of the process is essentially precise in nature but randomly distributed over an outcome space.
- The **laws of probability** apply to variable quantities. Whether explicitly or implicitly, the concept of the **repeated experiment**, which is at the heart of statistical theory, can be applied to variability.

EPISTEMIC INTERPRETATION OF VARIABILITY

In ecology there is seldom a situation where experiments can literally be repeated. As pointed out by Thomas (1995), for one thing, time will have elapsed. In dynamic systems, such as ecosystems, this will mean that the system has already moved to another point in its state space, and that in principle, no experiment can be exactly duplicated. However, there may exist an **experimental morphology**, which, for the observer's purposes, is repeatable.

> Example: Thomas (*op. cit.*) quotes the mathematician Cramer in describing the assessment of the probability in 1944 that the Second World War would come to an end. Although this war was unique in history, there were elements with regard to the strategic positions of the various armies, the morale of the troops, the resources available to the warring factions etc., that could be compared to those in other conflagrations, and which would lead the observer to estimate the likelihood of an end to hostility.

Table 2.1. Some of the characteristics of uncertainty and variability with particular reference to ecological models (based on Frey, 1993 and USEPA, 1997b).

Characteristic	**Uncertainty**	**Variability**
Source	Lack of empirical knowledge of the observer or imperfect means of observation.	True heterogeneity inherent in a well characterised population
Impacted by:	Model uncertainty • Model structure • Range of conceptual models Parameter uncertainty • Random error due to imperfect measurement • Systematic error (bias) • Inherent stochasticity or chaos • Lack of empirical basis • Unverified correlation among uncertain quantities • Expert disagreement on data interpretation	Individualism in response Lack of representative data Aggregation dimension (e.g. time or space)
Encoding	(Bayesian) Probability distribution	Frequency distribution
Effect of more data	Reduces	Unchanged but more precisely known
Applicability of standard statistical data analyses	Understated (due to focus on random error to the exclusion of bias introduced by variability)	Overstated (due to inclusion of measurement error)

2.3.2 UNCERTAINTY, VAGUENESS AND AMBIGUITY

UNCERTAINTY

It is necessary to distinguish between uncertainty and variability since it has an impact on the way in which likelihood is expressed and interpreted. The likelihood of a phenomenon of the model may be influenced by two broad categories of causes: epistemic uncertainty or systemic uncertainty.

- **Epistemic uncertainty** refers to the situation where the knowledge about, and hence the description of the system is uncertain

- **Systemic uncertainty** refers to the situation where the system itself is uncertain in its definition even though the tools for its description are precise. A comparison between uncertainty and variability is made in Table 2.1.

Essentially, what distinguishes uncertainty from variability is the lack of a stochastic basis. Uncertainty is a characteristic of an observer rather than of a system and stems from a lack of knowledge. Frey (1993) resolves two kinds of uncertainty: model uncertainty and parameter uncertainty.

- The model uncertainty in the case of ecosystem models is due to imperfect knowledge of a specific ecosystem's processes and mechanisms. There may be several options that may be conceptually valid based on the study of other similar ecosystems or mechanistic models.
- The stress responses may be quite precise, but the discrimination among the model choices may be blurred. This phenomenon is exacerbated by parameter uncertainty. Even when the specific model used to predict effects is known, very often the parameter values are wholly or partially unknown or the number of parameters is unknown. The sources of parameter uncertainty are listed in Table 2.1. It is apparent the variability as used above may be a subset uncertainty.

In many cases, it is possible to extrapolate from simple systems, such as laboratory test systems, to ecosystems on various bases, but with a significant loss in confidence (See Table 2.2). However, much of the work done on extrapolation and projection is only applicable to the effect of toxics. Characteristic of these extrapolations is the dependence on system specific knowledge and the rapid increase in uncertainty.

VAGUENESS AND AMBIGUITY

In the description of variability and uncertainty in Section 2.3 above, the outcome of stress is precise although not deterministically predictable. In principle at least, an experiment can be conducted which will elucidate the effect of a stressor on an individual organism (for example) and that will uniquely define that particular individual's response. Repeating the experiment on a large number of individuals will characterise the expectation of response better but it will not remove the variability of the population response.

In contrast to variability, the observer's personal sense of confidence in assessing the outcome of stress applied to an ecosystem may also be hampered by uncertainty, vagueness and ambiguity. These differ from variability in that, while variability is a characteristic of the system, uncertainty, vagueness and ambiguity is a characteristic of the observer.

In contrast to uncertainty, vagueness relates to the precision with which inputs and outputs in the predictive or analytical process is known. In the context of the NWA, terms such as "sustainable" are left undefined. The definitions in 2.4.2 derived from literature sources, are vague. In addition, qualifiers such as "adequate sustainability", "adequate resilience" and "massive abnormal mortality" are functionally vague terms but are nevertheless descriptive. The choice of phraseology is intentionally vague as the values by which it is characterised is highly site- and situation-specific. A term such as "adequate" as a qualifier for sustainability may take on a range of values as opposed to the qualifiers "low" or "high". But the interpretation of the term is qualitatively clear and its implications scientifically interpretable.

2.4 STRESSOR RESPONSE RELATIONSHIPS FOR ERBM

As noted in 2.2 above, a SRR is a functional relationship between an end-point and the magnitude of the stressor. In view of the impact of uncertainty and variability as discussed in 2.3, it may in general be impossible to specify ecological effects deterministically. Consequently, an ecological SRR may at best be expressed as a likelihood that a selected endpoint may be observed. For ERBM decisions to be scientifically tenable and legally valid, the SRR should:

a. Refer to the regulatory end-point rather than a laboratory or other field observational end-point (i.e. the Response Inference problem referred to in 2.4.1), and
b. Make the best possible use of all relevant information. This involves formulating the Response Inference on a basis suitable to the data at hand (2.5 and 2.6).

2.4.1 THE RESPONSE INFERENCE PROBLEM

The general form of this problem can be described as follows: "You (the assessor) are required to make a pronouncement about the impact of a stressor at a higher level of organisation (such as at the ecosystem level) and at a conceptual level (in terms of sustainability for example) which is far removed from the experimental data You have available". The problem, therefore, concerns both organisational and conceptual scaling of response end-points.

THE ISSUE OF SCALE

Figure 2.5 illustrates the problem with scale in the estimation of ecological stressor-response relationships. The difference in scale results in an incongruence between the level of the data available for making decisions and the level of the impact of those decisions.

Data scale

In many cases estimates of effect are based on laboratory data generated from experiments performed to observe the change in physiological functions of individual organisms (e.g. measured as change in reproductivity, cessation of vital function, change in behaviour, etc.) on exposure to a stressor. It estimates effects at a scale of perhaps a few millimetres to perhaps tens of metres (in the case of micro- or mesocosm experiments) and hours to perhaps a few months (Sugiura, 1992; Graney, *et al.*, 1994). The regular experiments may therefore cover the domain of spills or short-term pollution incidents.

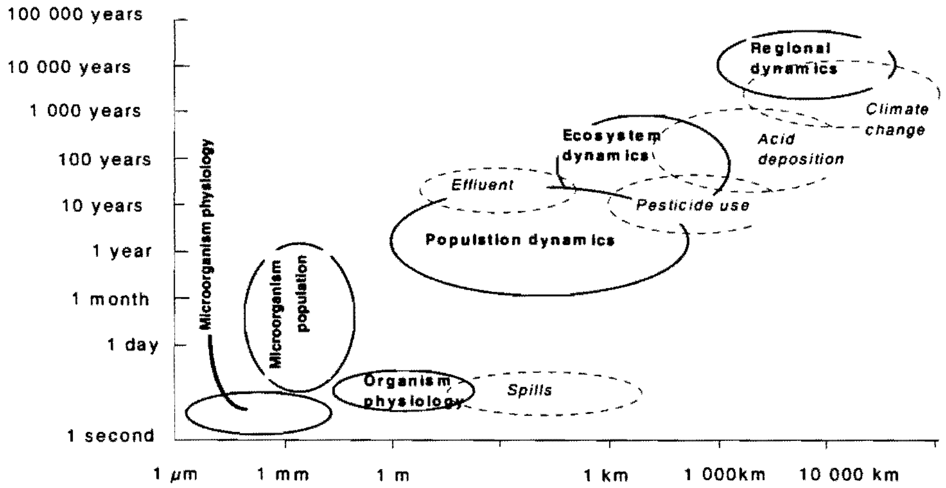

Figure 2.5 Temporal and spatial domains of some ecological factors and typical stressors (adapted from Suter, 1993).

Management scale

The greater problem for South African surface water management, where the major source of flow in the dry season is comprised of effluent, is that its impacts occur in the spatial domain of tens of meters to several kilometres and the temporal domain of several decades.

Regulatory scale

The National Water Act sets a goal (ecological sustainability) at ecosystem scale, for which the responses are in the spatial domain of a few meters to hundred of kilometres in the temporal scale of a few years to centuries. In many cases it is not possible to conduct experiments at the required spatial and temporal scale to estimate stressor response directly. There is a need to perform extrapolations from the observational scale to the required scale (Suter, 1990; Landis and Yu, 1995). When extrapolations such as those in Table 2.5 (Section 2.7) are used, it should be carefully noted whether the extrapolations refers to both spatial and temporal scaling.

Scaling impacts or responses over different levels of ecological organisation, spatial and temporal domains necessarily means that there is a loss in confidence. To address this uncertainty systematically, a model of relationships of various end-points pertinent to the aquatic ecosystem is needed.

2.4.2 ECOLOGICAL PHENOMENA

A distinction is now made between issues (such as "sustainability", "integrity", etc.) and end-points, which specifies some characteristic of the issue (such as "loss of sustainability"). It is proposed that when higher level issues, such as sustainability are addressed, there are natural "milestone issues" that can be defined in terms of biological descriptors such as "integrity", "biodiversity", etc. These issues can be associated end-point events or phenomena, which would be described as the attainment (or conversely, the loss) of such a "milestone event".

In an assessment of risk at this level, the term "likelihood" essentially expresses confidence that such an event can (or has) taken place. Each phenomenon or event may, in principle, be arrived at in many mechanistically different ways, each of which influences the likelihood that the phenomenon could be observed. However, the likelihood of observing a phenomenon is not dependent on knowledge of the mechanistic detail, but rather on the epistemology of the event.

A phenomenological rather than a mechanistic basis is chosen to facilitate the incorporation of expert judgement and observational data at higher levels of ecological organisation (where mechanistic knowledge is often lacking). It is assumed that a phenomenological model should have the following characteristics:

A. The phenomena should be linked by logical inference.
B. Methodology should be available to assess the state of the phenomena, which implies that there should be metrics for the state (e.g. see Table 2.2). The risk is then the expression of

the likelihood that a given set of state-descriptors characterising the phenomenon is attained or lost.

C. The phenomena should be chosen at an organisational level suitable to the assessment (Figure 2.5). As the state of mechanistic knowledge increases, the phenomena could be resolved further until, conceptually, phenomena at molecular level or lower can be related to the higher level phenomenon. If no measurement end-point exists at the level of the assessment, the assessment should not be changed to suit the end-point. Rather the model should be used to emphasise the information need. Failure to do this results in a false sense of confidence.

Table 2.2 Indicator variables for assessing biodiversity at three levels of organisation. (Based on Noss, 1990 and augmented from Pratt and Cairns, 1996, Karr, 1993)

Level	**Indicators**		
	Composition	**Structure**	**Function**
Community/ecosystem	Identity, relative abundance, frequency, richness, evenness, diversity of species or guilds, succession	Abundance, density and distribution of key physical features and structural elements of rivers, Food web assembly	Biomass productivity, parasitism, predation rates, colonisation end local extinction rates, patch dynamics, nutrient cycling rates, biogeochemical cycles
Population/species	Absolute or relative abundance, biomass, density, primary production and primary and secondary consumption	Dispersion (micro distribution), range, population structure (e.g. age ratio), habitat variables (as above)	Demographic processes (e.g. fertility, survivorship), population fluctuations, physiology, life history, individual growth rates
Genetic /cellular	Allelic diversity	Census and effective population size, generation overlap, heritability	Inbreeding depression, gene flow, mutation rate, selection intensity, and photosynthesis.

MODEL POSTULATES

The conceptual model is based on the following postulates:

1. The **reference state** for the model is the **pristine system**. It is implicitly assumed that the reference state's only fixed characteristic is that it is pristine, but that the values of the descriptors could be spatially and temporally variable. There exists a pristine **pattern of natural extreme events** such as droughts or floods which are not stressful and which may be necessary (due to adaptation) in arid or semi-arid regions such as South Africa (DWAF, 1987; Davies, *et al.*, 1994).

2. The quest for the maintenance of sustainability only arises because there is real or implied **anthropogenic threat** to the system. Sustainability is not defined for a system not subject to any threat of anthropogenic stress.
3. The phenomenon "sustainability is maintained" occurs only if the phenomenon " suitable level of integrity is maintained" occurs. The state of **integrity** of the system is determined by its **state of biotic integrity, habitat integrity and the natural temporal patterns of extreme events**. For integrity to be maintained neither habitat diversity, nor biodiversity nor the natural temporal event pattern should have been disrupted (Odum, 1985; Pratt and Rosenberger, 1993; Naeem, *et al.*, 1994).
4. **Biodiversity**, in terms of the **composition, structure** and **function** of the system (each at several levels of organisation from molecular to landscape level) is defined in relation to the state of these components in a pristine system. Biodiversity as a variable indicating stress is subject to an interpretation of the **individual importance** of species. Redundancy is possible or even probable in an ecosystem and the real question is how much diversity could be lost without pushing the system to the edge of some irreversible, catastrophic change (DeLeo and Levin, 1997). The conservative assumption would be that **all species are equally important** (rivet popper hypothesis) (Walker, 1991).
5. For biodiversity to be maintained, neither the structure nor the function of biota should have been impaired. Any such **impairment**, by definition, implies **loss of integrity**.
6. Rapport, *et al.* (1985) point out that integrity is lost more easily in a system subject to **constant low-level stress** compared to a system subject to **infrequent high intensity stress**. Qualitatively this is modelled analogous to the model of reversible toxic effect (e.g. Hathway, 1984; Verhaar *et al.*, 1999; Freidig *et al.*, 1999). The absence of stress is interpreted to mean that, either or both the level of the stressor was not high enough, OR that the duration of exposure to the stressor was too short to make any impact.
7. An ecosystem is assumed to be impacted by **chemical water quality** or **physical quality** of its **habitat**, or by the stress related to the **flow rate** of the water comprising its physical habitat or by the presence of **exotic biota**.
8. The long-term effect of stressors is also dependent on the **availability of refugiae** from which the population numbers can be replenished. If no such refugiae exist, then the **population viability** is dependent on **sufficient numbers** to maintain its status despite natural mortality and normal biotic interactions such as predation and competition. The **precautionary approach would be to assume that no refugiae exist**, but this restriction could be lifted on a site-specific basis.

2.4.3 A CONCEPTUAL MODEL FOR END-POINT SCALING

PHENOMENOLOGICAL INFERENCE

The laboratory-level observations are linked to the conceptional level end-point by induction on the phenomena (*sensu* Thomas, 1995). Induction relies on the modeller's conception of how the various concepts are linked to one another, and how the concepts are linked to the material world. If A and B are phenomena at different organisational and conceptual levels, then the question "If the knowledge of the sate of A changes, will it impact on the sate of B" has to be repeated for all the phenomena under consideration. This implies that a system analytical model of the interactions be constructed based on the current insights on the system.

(a) As a first step a diagram as shown in Figure 2.2 might be generated where the direction of the arrows indicates the direction of influence. This also means that with equal validity a different conceptualisation will lead to a different model.

(b) The next step is to quantify the influence relationships. This would involve a) the quantitative or qualitative change in one state of one phenomenon as a function of the change in state of another phenomenon/phenomena, and b) the strength of that relationship.

INFERENCE MODEL STRUCTURE

Return now to the problem of estimating the likelihood of sustainability (or more precisely the unsustainability, which is defined largely at a conceptual level) based on current knowledge and observational data. It is necessary to link current understanding of ecosystem concepts to the stressors that are to be managed in such a way that finally the likelihood of ecosystem sustainability is expressed as a function of stressor characteristics.

The idea is to encapsulate system knowledge in a rule base expressing the relationships between phenomena (ρ). If ρ is combined with the site-specific evidence base (ε) in the form of a conjunctive combination, $\rho \wedge \varepsilon$ (where $\wedge$ indicates "conjunction"), then the outcome of this operation expresses the conclusion regarding the system status.

The rule base ρ can be rewritten in the canonical form to illustrate how it can be combined with the evidence ε in the two most often used forms of reasoning, the *modus ponens* and the *modus tollens* (DuBois and Prade, 1988).

Modus ponens:

Rule (ρ):	If V is A_1 then U is B	
Observation (ε):	<u>V is A_2</u>	[2.1]
Conclusion ($\rho \wedge \varepsilon$):	U is B'	

Modus tollens:

Rule (ρ):	If V is A then U is B1	
Observation (ε):	U is B2	[2.2]
Conclusion ($\rho \wedge \varepsilon$):	V is A'	

Step 1: Constructing an influence diagram

By repeatedly applying a *modus ponens* or *modus tollens* reasoning, a conclusion can be drawn regarding the truth of the antecedent.

From the postulates and the inference rule base in Section 2.4.1, a typical "fault tree" type of diagram can be constructed as shown in Figure 2.6. This is the basis of the phenomenological model.

Step 2: Quantifying the influence relationships

Applying this format (Eqs. [2.1] and [2.2]) to the postulates and the rule base in the appendix yields expressions like Eqs. [2.3] to [2.7] below.

Rule VIa :	$lc1 \wedge dc0 \rightarrow Cmps$	(η_6 true) [1]	
Observation:	lc1	(α true)	[2.3]
Observation:	dc0	(β true)	
Conclusion:	Cmps	(γ true)	

Rule Va:	$Cmps \rightarrow \neg Cmp$	(η_5 true)	
Observation:	Cmps	(γ true)	[2.4]
Conclusion:	$\neg$Cmp	(χ true)	

Sus: Sustainability is assured, *Res*: Resilience is assured, *Int* : Integrity is assured, *Div*: Biodiversity is intact, *Tpat*: Temporal stress/recovery patterns are undisturbed, *Cmp*: System composition is undisturbed, *Str*: System structure is undisturbed, *Fct*: System function is normal, *Tpats*: Temporal stress/ recovery patterns are in a state of stress, *Cmps*: System composition is under stress, *Strs*: System structure is under stress, *Fcts*: System function is under stress, *lxi0*: Minimally significant level of stressor *X* exists for integrity component *i*, *dxi0*: Minimally significant duration of exposure to stressor *X* exists for integrity component *i*, *dxi* : Long duration of exposure to stressor *X* exists for integrity component *i*, *lxi*: Intense exposure to stressor *X* exists for integrity component *i*, where $X \in$ {toxic substances (*T*), flow deficiency (*Q*), nutrient disruption (*N*), system driving variables disruption (*S*), physical habitat disruption (*H*)}, and $i \in$ {*Cmp* (*c*), *Fct* (*f*), *Str* (*s*), *Tpat* (*t*)}.

[1] $\rightarrow$ indicates logical implication and $\neg$ indicates "not" or logical negation

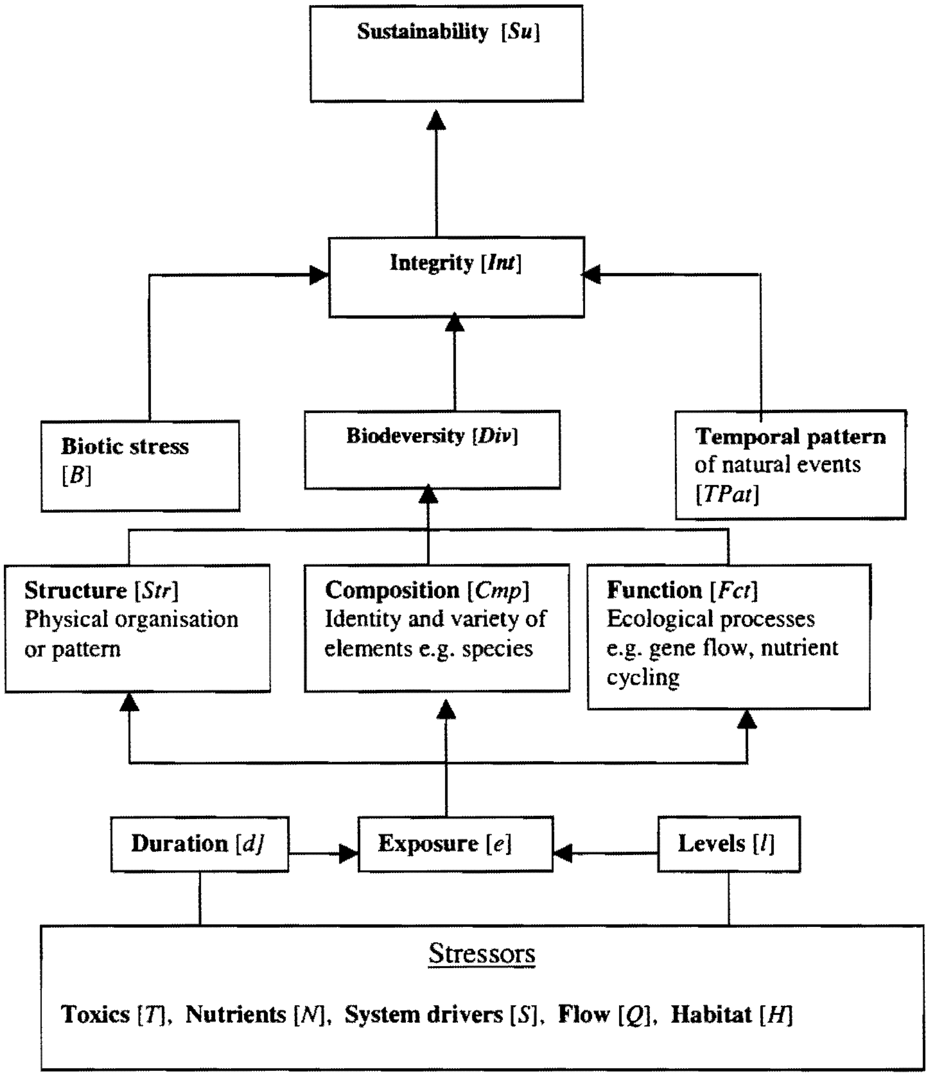

Figure 2.6. A systems model for ecological effect inference. The boxes indicate phenomena. The direction of the arrows shows the direction of influence. The blocks in Figure 2.6 must be read in conjunction with necessary characteristic, e.g. "Integrity" should be read: "The phenomenon of attainment of suitable integrity".

Rule IV:	$\neg$Cmp $\rightarrow$ $\neg$Div	(η_4 true)	
Observation:	$\neg$Cmp	(χ true)	[2.5]
Conclusion:	$\neg$Div	(δ true)	

Rule III:	$\neg$Div $\rightarrow$ $\neg$Int	(η_3 true)	
Observation:	$\neg$Div	(δ true)	[2.6]
Conclusion:	$\neg$Int	(ε true)	

Rules I and II:	$\neg$Int $\rightarrow$ $\neg$Sus	(η_2 true)	
Observation:	$\neg$Int	(ε true)	[2.7]
Conclusion:	$\neg$Sus	(τ true)	

Similar reasoning can be used for all the stressors and for other elements of biodiversity. Finally a conclusion can be reached as to the truth-value of the end-point: (e.g. "It is largely true that sustainability will be maintained" or " It 85% probable that the system will maintain > 95% sustainability").

2.4.4 DERIVING A RISK EXPRESSION FROM THE INFERENTIAL RULE BASE

Eqs. [2.3] to [2.7] express the inference of the system sustainability from the characteristics of the stressors occurrence. However, this inference is not yet a risk measure. It should be recalled that each of these inferences is the subject of an observer's conception induced onto perceptions of phenomena and that there is a measure of uncertainty in each inference. If it is supposed that the uncertainty can be described by a likelihood measure, Λ, that expresses an observer's (or a body of observers') confidence in the inference, then the measure of likelihood, $\Lambda(\neg Sus)$ is a risk measure. Each of the inferences can be represented by a conditional likelihood of the form: if A $\rightarrow$ B the uncertainty in the inference can be assessed by $\Lambda(A \mid B)$, i.e. the conditional likelihood of A given B. The exact form of the reduced likelihood depends on the measure Λ. Two types of likelihood measure are commonly used; each based on a different logic and each with its own calculus:

1) If the underlying **logic is crisp** (i.e. each proposition in the rule base is either true or false and nothing else, i.e. the values η_2 to $\eta_6 \in \{0, 1\}$ where 0 denotes "false" and 1 denotes "true") then results of probability theory are applicable and, consequently, Λ is then a **probability** measure and the results would belong to the domain of probabilistic risk assessment.
2) If the underlying **logic is fuzzy**, i.e. the values η_2 to $\eta_6 \in [0, 1]$, then the results of possibility theory are applicable and, consequently, Λ could be any one of a number of **possibility measures** each with a different interpretation and the risk will be possibilistic. Many of the phenomena (such as the existence of integrity) are essentially vague, and it is likely to benefit from a fuzzy approach.

Interpretation of the terms "risk", "probability" and "possibility" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997). The interpretation of likelihood is crucial to decision-making in data-poor ecological management situations.

2.4.5 SET-THEORETICAL DESCRIPTION OF THE INFERENCE PROBLEM

A set theoretical description is used to illustrate the point. Assume, for example, that the phenomenon "Ecosystem sustainability is lost" is used as an end-point. It is known that an infinite number of combinations of stressor states can result in this phenomenon. Assume that all the combinations of stressor states that correspond to the event: "sustainability is lost" are assembled in a set.

In defining the end-point phenomenon, the questions now arise: "At what point or combination of events can it be said that ecosystem sustainability is 'lost'? Is there a specific point at which it can be said that sustainability is lost? Or is there rather an increasing confidence in describing the system as being unsustainable?" The answers to these question can be summarised as in Table 2.3.

Table 2.3 The assessment of the state of the end-point phenomenon (loss of sustainability) and the state of lower level phenomena.

Case	**End-point phenomenon (set boundary)**	**Component phenomena (elements of the set)**	**Interpretation**
A	Crisp	Crisp	There exists a clearly defined set of threshold values that define a unique point representing system unsustainability.
B	Fuzzy	Crisp	Although the component events are clearly defined, the state corresponding to system collapse is vague.
C	Crisp	Fuzzy	The point of collapse is clearly defined but is not known how or when that state is reached.
D	Fuzzy	Fuzzy	Neither the point of collapse nor the threshold values are clearly defined.

The answers to these questions clearly lend different interpretations to the term risk since the likelihood that a parameter vector belongs to this set defines the risk.

If A and C are true it may still be that the parameters are subject to stochasticity. In this case risk is interpreted as the likelihood that a particular parameter vector of event states will belong to the set or not. Likelihood can be described in terms of probability theory, which requires a definable event to activate its precepts. In contrast to the frequentist view of probability, where probability is a limiting value of a series of repeated observations, the Bayesian view, where probability characterises the observer's sense of expectation, based perhaps on morphologically similar situations, can be used.

At the other end of the conceptual spectrum is the situation where B and D are true. The likelihood cannot be expressed in terms of the *probability* that a parameter vector belongs the set because the set and its elements are ill defined. The only recourse is to express likelihood in terms of fuzzy set theoretical likelihood measures such as possibility and necessity.

2.5 PROBABILISTIC FORMULATION OF THE END-POINT INFERENCE PROBLEM

In the literature referenced in this study, wherever risk is characterised quantitatively, the likelihood is expressed in terms of probability. Interpretation of the terms "risk" and "probability" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997) and particularly to decision-making in data-poor ecological management situations.

2.5.1 PROBABILITY THEOTERICAL APPROACH

Two approaches to a probabilistic expression of likelihood can be distinguished:

- The "frequentist" approach (Jaynes, 1996), sees probability as the limiting frequency of an occurrence over a large number of observations.
- In contrast, probability can be seen as a subjective expression (not necessarily dependent on repetitive observations) needed to project from the domain of uncertainty by the means of prevision to the domain of certainty. "Prevision, ... consists in considering, after careful reflection, all the possible alternatives, in order to distribute among them, in the way which will appear most appropriate, one's own expectations, one's own sensations of probability" (DeFinetti, 1990). With this view in mind, probability, and by association risk, could be seen as epistemic of the specific combination of situation and assessor.

Regulatory decision-making in the field of ecology is characterised by:

- ✓ A descriptive conceptual knowledge of ecosystems, often only supported by patchy observation.
- ✓ Observations of multiple replicates of experiments are often not available or simply impossible. The only recourse is then to expert prevision pertaining to a specific situation.

This is still in keeping with the principle of risk assessment. Predictive ecological risk is essentially an expectation of an effect, a prevision based on best available knowledge of the assessor's knowledge of and expertise in dealing with, what are as yet, unobserved events in a complex system. The calculated ecological risk values are therefore an expression of the assessor's expectation, taking into consideration the scientific information at hand.

In this section the expression of likelihood as probability is considered. (Note: Likelihood is not be confused with "likelihood" or "likelihood ratio" used in Bayesian statistics.) In expressing the uncertainty about the inferential expressions in the model, the use of probability theory was mentioned in respect of the use of binary or Boolean logic.

UPPER-LEVEL PHENOMENA

For those phenomena that are naturally concerned with levels of ecological organisation above that of population, the crucial inferences are Eqs. [A2.7] to [A2.9] in the Appendix

$$\neg(Cmp \wedge Str \wedge Fct) = \neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div \qquad [A2.7]$$

$$\neg(Div \wedge Tpat) = \neg Div \vee \neg Tpat \rightarrow \neg Int \qquad [A2.8]$$

$$\neg Int \leftrightarrow \neg Sus \qquad [A2.9]^2$$

Each of the elements (Cmp, Str, Div, Sus, etc) refers to an end-point phenomenon that is considered relevant to a specific ERA or ERBM situation.

Given the uncertainty in both the arguments and the inferences, the probabilistic ecological risk would mean that Eqs. [A2.7 to [A2.9] need to solved by application of Eq.[2.8] which refers to generic events p and q and probabilities a and b (Dubois and Prade, 1988) to yield the set of equations [2.9].

$$\begin{array}{l} P(p \rightarrow q) \geq a \\ \underline{P(p) \geq b} \\ P(q) \geq ab \end{array} \qquad [2.8]$$

$$\begin{array}{l} P(\neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div) \geq \eta_3 \\ \underline{P(\neg Cmp \vee \neg Str \vee \neg Fct) \geq \beta} \\ P(\neg \mathrm{Div}) \geq \eta_3\beta \\ \\ P(\neg Div \vee \neg Tpat \rightarrow \neg Int) \geq \eta_2 \\ \underline{P(\neg \mathrm{Div} \vee \neg \mathrm{Tpat}) \geq \alpha} \\ P(\neg Int) \geq \alpha\eta_2 \\ \\ P(\neg Int \leftrightarrow \neg Sus) \geq \tau \\ P(\neg Sus) \geq \tau\alpha\eta_2 \end{array} \qquad [2.9]$$

[2] $\leftrightarrow$ Denotes "if and only if" or logical equivalence.

If phenomenon p is considered logically equivalent to phenomenon q (i.e. $p \leftrightarrow q$) it is tantamount to asserting that one's knowledge of the uncertainty of the occurrence of p is no different from ones knowledge of the uncertainty of q and therefore $P(p) = P(q)$. However, the confidence in, or strength of the relationship (a in Eq. [2.8]) expressed as $P(p \leftrightarrow q)$ still needs to be assessed.

The probability of conjunction of phenomena in Eq. [2.9] may be difficult or impossible to assess. That would mean having knowledge of any of the endpoint phenomena occurring while the data at hand may only refer to the occurrence of phenomena in isolation. Consequently it is necessary to resolve the conjunction in terms of the probability of occurrence of individual end-point phenomena. The partitioning of a composite event probability into component event probabilities is accomplished by Eq. [2.10] (DeFinetti, 1990) where an event E is partitioned into n different logically independent events E_i where $i \in \{1,2,\ldots,n\}$. :to the conjunctions in Eq. [2.9] to the set Eq. [2.11].

$$P(E) = P\left(\bigcup_{i=1}^{n} P(E_i)\right) = \sum_{i} P(E_i) - \sum_{i \neq j} P(E_i E_j) + \sum_{i \neq j \neq k} P(E_i E_j E_k) - \ldots \pm P(E_1 \ldots E_n) \qquad [2.10]$$

Eq. [2.10] now contains terms that require the probabilities of conjunctions. These may be even less well known in an ecosystem context than the corresponding disjunctions. However, if one were to assume that the end-point phenomena are independent (i.e. that one's knowledge of the occurrence of one end-point in the conjunction is independent of one's knowledge of the occurrence of the other end-points), then the probability of the conjunction becomes the product of the individual phenomena probabilities.

Furthermore, analysis of Eq [2.10] (with the assumption of independence included) shows that Eq. [2.11] will always be true.

$$\max_i \{P(E_i)\} \leq P\left(\bigcup_i E_i\right) \leq \min_i \left\{\left(\sum_i P(E_i)\right) 1\right\} \qquad [2.11]$$

If the individual phenomena probabilities are known:

$P(\neg Cmp) = \beta_1$, $P(\neg Str) = \beta_2$, $P(\neg Fct) = \beta_3$ and $P(\neg Tpat) \geq \alpha_1$

Then

$\beta = \max\{\beta_1, \beta_2, \beta_3\}$

$\alpha = \max\{\alpha_1, \eta_3\beta\} = \max\{\alpha_1, \eta_3\beta_1, \eta_3\beta_2, \eta_3\beta_3\}$

$P(\neg Sus) \geq \tau\alpha\eta_2 = \max\{\tau\eta_2\alpha_1, \tau\eta_2\eta_3\beta_1, \tau\eta_2\eta_3\beta_2, \tau\eta_2\eta_3\beta_3\}$ [2.12]

Up to this point only higher-level phenomena had been addressed. A connection between higher and lower (laboratory-level) phenomena is proposed in Appendix 2.10.3. The combination of higher and lower level phenomena is shown diagrammatically in Figure 2.7.

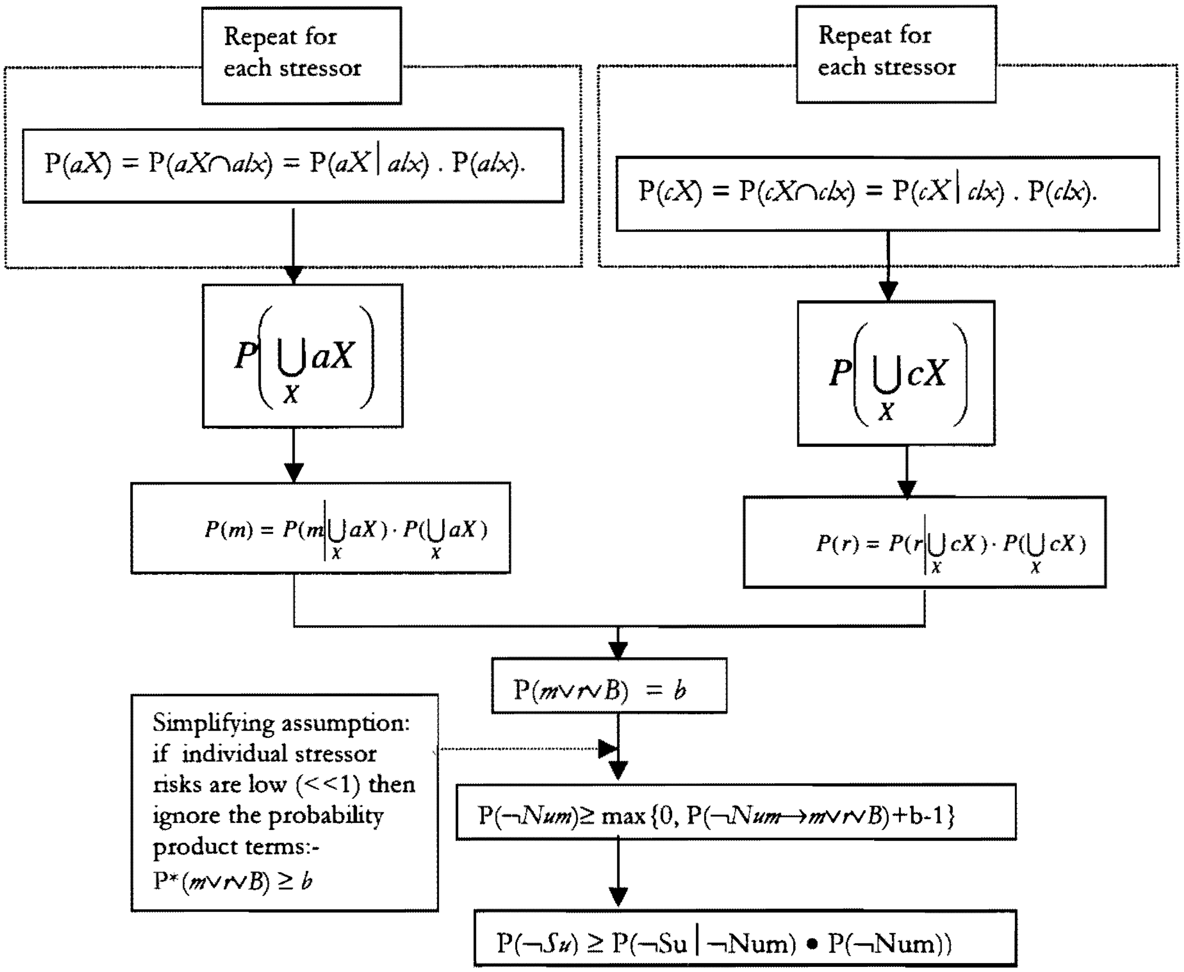

Figure 2.7 A diagrammatic representation of the process for estimating the confidence in high-level end-points from low-level (e.g. laboratory-level) end points.

2.6 POSSIBILISTIC FORMULATION OF THE END-POINT INFERENCE PROBLEM

2.6.1 BACKGROUND TO FUZZY APPROACH

The concept of fuzzy sets is commonly ascribed to the early work by Zadeh (1965). The essential difference between fuzzy and classic (crisp) sets lies in the definition of the sets. For crisp sets the universe of discourse is dichotomised into those events that belong to the set and those that do not (Klir and Folger, 1988), i.e. there must be a bijection between the sample space and the event space (Dubois and Prade, 1988). A probabilistic model is suitable for precise but dispersed information. In many real life complex situations this type of distinction is not that easy to make.

Each event is assigned a degree to which it is perceived to belong to the set under discussion (degree of membership μ).

2.6.2 THE RATIONALE FOR A FUZZY APPROACH TO RISK

Possibility theory (based on fuzzy set theory) (DuBois and Prade, 1988) may be better suited to the kind of situation where semi-quantitative expert opinion, such as in ecology, is the basis of the decision-making process. A fuzzy mathematical approach to ecological risk has been used (e.g. Ferson and Kuhn, 1993; Ferson, 1994) and possibility theory merits investigation as a total risk estimation tool.

Ecosystem characteristics

Some ecosystem characteristics could be interpreted at both a phenomenological and a mechanistic level. Concepts such as sustainability and resilience may be spatially and temporally scale dependent and the knowledge of the mechanisms underpinning these phenomena are vague (Costanza *et al.* 1993, De Leo and Levin, 1997). However, changes in the state of these phenomena are observable. As an example of the complexity of the mechanics related to such phenomena, is the natural variability and successional cycling in a system, which drives many of the ecosystem processes. If these are disrupted, a system may be produced that is structurally different to the original system. "Therefore, in managing ecosystems, the goal should not be to eliminate all forms of disturbance, but rather to maintain processes within limits or ranges of variation that may be considered natural, historic or acceptable" (De Leo and Levin, 1997).

Not only natural variability has to be accounted for in the management process, but also uncertainty and in some cases vagueness. Some definitions of ecosystem integrity; e.g. "the maintenance of the community structure and function characteristic of a particular locale or deemed satisfactory to society" (Cairns, 1977) or "the capability of supporting and maintaining a balanced, integrative, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley, 1981), although epistemic, is essentially vague and subjective. The system boundaries, the response to stressors and the stressors themselves may only be known qualitatively. The functional entities that best reflect the goals of ecosystem management may only be vaguely identifiable. Consequently, in dealing with ecological risk in the context of protective ecosystem management, it would be advantageous to use a paradigm that is adapted to address both uncertainty and vagueness such as possibility theory, which is based on the use of fuzzy logic.

Fuzziness in the inference model

The response inference model of (Appendix Eqs. [A2.1] to [A2.5d]), is essentially based on inference of form (*sensu* Thomas, 1995) rather than content. Eqs. [A2.4] to [A2.7d] and [A2.8a] to [A2.11] are expressions based on the formalisms of Aristotelian logic. If the assertion: A→B is made, this was essentially accepted as being true or false. In the probabilistic formulation of the inference model in Section 2.5, it was assumed that, due mostly to variability, there was a certain probability that this implication was either true or false. The only source of the uncertainty in this case was the variability in individual responses (stress) to stressors and the variability in exposure of the target entities to stressors. Consequently, the unique identification of both target entities and end-points for assessment was considered crucial.

However, if the definitions of sustainability, resilience and integrity, are considered, it becomes clear that it is not that easy to define target entities such as the ecosystem or what exactly is meant by "compromised sustainability", "loss of resilience", "compromised integrity", "corrupted composition", "abnormal system function", etc. There is an additional uncertainty imposed by vagueness in terminology that can only be eradicated by rigorous definition, which is unlikely to be mirrored in the precision and extent of the knowledge base or the definition of the system boundaries.

Moreover, it is likely that measures such as normality and integrity would be interval valued rather than single valued. All the assessments in the rule base may have to be made with reference to the condition of being intact or pristine. With an uncertain (fuzzy) knowledge base the assessment of *Fct* and *Int*, for example, would generally be of the type: "Largely normal" or "significantly impaired". However, the condition of being "undisturbed" is difficult to establish, but an observation about the system may to a greater or lesser degree be said to correspond to the condition of being "undisturbed". This means that both antecedents and consequents in Eqs [2.3 to 2.7] are fuzzy quantities. This places a suitable model for end-point projection in either Cases B or D of Table 2.3, but most likely in Case D.

A FUZZY INTERPRETATION OF THE ECOLOGICAL INFERENCE MODEL

In general, the rules on which the inferences are based are of the form "If $\widetilde{X}$ is $\widetilde{A}$ then $\widetilde{Y}$ is $\widetilde{B}$" where $\widetilde{X}$, $\widetilde{A}$, $\widetilde{Y}$ and $\widetilde{B}$ are generally vague. Recall that the propositions on which these rules in the Appendix were based refer to the pristine state. Rule I (See Appendix) could then be expressed alternatively as " The assurance of sustainability of the system takes its value from the (fuzzy) set of pristine values". It seems unlikely that the value of assurance of sustainability could have been given a specific value that would have been measurable and which could have been given a numerical value, since sustainability is merely a concept. At best the adequacy of the system sustainability could have been described as "very high" in a pristine system.

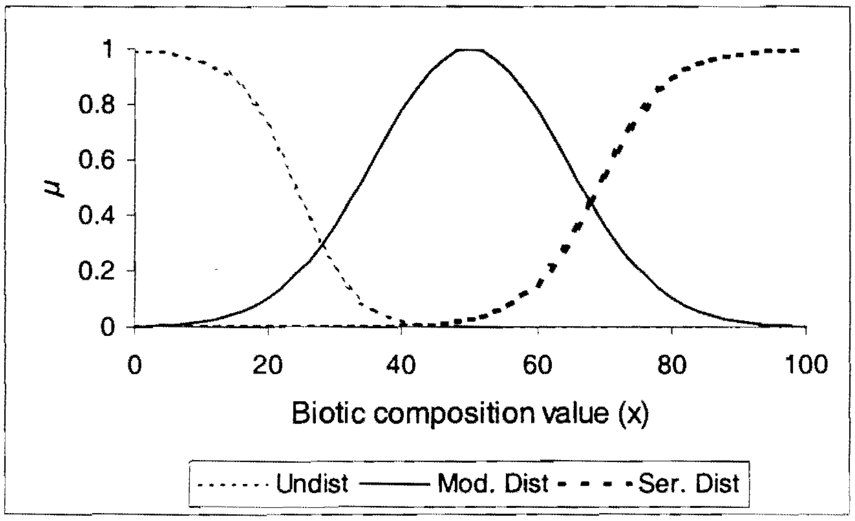

Figure 2.8. Example of a relationship between the value of a hypothetical biotic composition variable x and the degree to which it corresponds to the description "Undisturbed", "Moderately Disturbed" and "Seriously Disturbed". It is assumed that $x \in [0, 100]$ with 0 indicating a definitely seriously disturbed condition and 100 indicating a definitely undisturbed condition.

This degree of correspondence to the state of being pristine is expressed by the membership function μ of an observation x with respect to the set *Cmp* and is expressed as: $\mu_{Cmp}(x)$. In principle a variable x related to system biotic composition can be evaluated and a curve set up that relates the value of x to the degree with which it corresponds to the state of being undisturbed (i.e. $\mu_{Cmp}(x)$). The qualifier "undisturbed" might also be replaced with "mildly disturbed" or "seriously disturbed". This will give rise to series of curves as shown in Figure 2.8.

2.6.3 POSSIBILITY THEORETICAL APPROACH

Fuzzy logic is better geared to handle the domain of vague premises and conclusions and, consequently, the likelihood operator, Λ (Section 2.4.4), can best be replaced by the possibilistic counterparts from the domain of possibility theory.

Analogous to the relationship of probability theory to crisp set theory is the relationship of possibility theory to fuzzy set theory. One of the features of the application of fuzzy set theory and possibility theory is the ability to use non-numeric quantifiers in computing. It is inherently able to deal with both numeric and non-numeric data. Probability theory has no means to distinguish a state of certain knowledge that a system is stochastic and the state of knowledge uncertainty about a deterministic event. Possibility theory makes this distinction (DuBois and Prade, 1988).

If x takes its value over V, and y takes its value over U, and furthermore if V and U are normalised sets (i.e. sets where $\exists\, x \in V$ such that $\mu_V(x) = 1$), then the rule and observation and conclusion can be formulated in terms of possibility distributions or membership functions (DuBois and Prade, 1988) for *modus ponens* and *modus tollens* as Eqs. [2.13 and [2.14] respectively.

$$\mu_{B'}(y) = \sup(\pi_{U|V}(x,y) * \pi_V(x)) = \sup[(\mu_{A1}(x) {*\!\rightarrow} \mu_B(y)) * \mu_{A2}(x)] \qquad [2.13]$$

$$\pi_A(x) = \mu_{A'}(x) = \sup[(\mu_A(x) {*\!\rightarrow} \mu_{B1}(y)) * \mu_{B2}(y)] \qquad [2.14]$$

where the operators * and *→ are dependent in the implication used as defined in Table 2.3

The inferential problem can be solved by determining the truth-value of $(A_2 \wedge (A_1 \rightarrow B))$. The conjunction is represented by the t-norm (T): $B' = \sup[T(A_2, (A_1 \rightarrow B))]$, where sup indicates the supremum over all the values over which A_1 and A_2 are evaluated.

*Table 2.3. The form of the fuzzy operator * (t-norm), the corresponding t-conorm and the fuzzy implication operator (*→) (Klir and Folger, 1988)*

Logic	a*b (t-norm)	t-conorm	a*→b
Gödel	Min(a, b)	max(a, b)	= 1 if a ≤ b = b if a > b
Goguen	a · b	a + b – ab	= 1 if a = 0 = min(1, b/a) otherwise
Lukasiewicz	Max(0, a+b-1)	max(a+b, 1)	min(1, 1-a+b)

The approach to characterising the truth-values derives from the observation that each of the inferential rules can be expressed as a conditional likelihood describing the confidence the assessor has in the veracity of the rule. The rules can also be rewritten as possibility distributions:

Rule I and II: $\Pi(Sus \mid Int) = \eta_2$

Rule III: $\Pi(Int \mid Div \wedge Tpat) = \eta_3$

Rule IV: $\Pi(Div \mid Cmp \wedge Str \wedge Fct) = \eta_4$

Rule Va: $\Pi(\neg Cmps \mid Cmp) = \eta_5$

Rule VIa: $\Pi(Cmp \mid (l1c0 \wedge d1c) \vee (l1c \wedge d1c0) \vee (l2c0 \wedge d2c) \vee (l2c \wedge d2c0)\ldots) = \eta_6$

Applying Eq. [2.13] to the set of conditions above yield the fuzzy truth value for the end-point ¬Sus:

$\Pi(\neg S_{us}) = \sup\{T(\eta_2, \varepsilon), T(\eta_3, \delta), T(\eta_4, \chi), T(\eta_5, \gamma), T(\eta_6, \alpha, \beta)\}$, where T indicates a suitable t-norm.

If the min operator is chosen as the t-norm, then the possibility of unsustainability as an end-point is given by Eq. [2.15].

$$\Pi(\neg S_{us}) = \sup\{\min(\eta_2, \varepsilon), \min(\eta_3, \delta), \min(\eta_4, \chi), \min(\eta_5, \gamma), \min(\eta_6, \alpha, \beta)\} \qquad [2.15]$$

CHOICE OF AGGREGATION OPERATOR

A number of *t-norms* and *t-conorms* have been developed in multi-valued logic and which are used to express intersection and union of fuzzy sets respectively. The most commonly used of these are listed in Table 2.3 (DuBois and Prade, 1988, Kruse *et al.*, 1994). The choice of these *t-norms* and *t-conorms* is not an implicit part of the process but have they to be explicitly chosen. Klir and Yuan, (1995) lists a number of axioms which could be criteria for the selection of operators. Two of those which may be particularly applicable to this model (in addition to the one above) and which stems from a requirement that fuzzy logic should collapse to Aristotelian logic, are:

- The equivalence of $a \rightarrow (b \rightarrow x)$ and $b \rightarrow (a \rightarrow x)$
- $a \rightarrow b$ is true if and only if $a \leq b$, i.e. fuzzy implications are true if and only if the consequent is at least as true as the antecedent.

2.6.4 APPROACH DEPENDENT RISK INTERPRETATION

A comparison of the interpretation of risk in probabilistic and possibilistic terms is given in Table 2.4. Risk expressed in **probabilistic terms** implicitly has the interpretation that if a similar set of conditions such as stressor exposure and stressor effect is observed often enough, the probability component of the risk will express the **number of times the end-point will be expected to be observed**.

On the other hand, with the **possibilistic (fuzzy) expression** of risk, an observer's description of the endpoint phenomenon will always have a sense of uncertainty irrespective of how many times a similar set of stressor states is observed. In the fuzzy interpretation, the risk corresponds to the observed or predicted **state corresponding to the notion of the end-point**.

The difference in interpretation can affect the "proveability" of risk. A probabilistic risk expression raises the possibility that if enough instances of identical stress are observed, the end-point effect will be observed because the end-point is ontologically certain. In contrast, the fuzzy risk expression is the result of epistemic or systemic uncertainty. Even if the expected end-point is not observed, each result observed under stress *similar* (but not necessarily the same) to that being modelled, will add to the evidence base, which either supports or rejects the risk characterisation.

Table 2.4. A comparison of the interpretation of risk in probabilistic and possibilistic terms.

Aspect of risk assessment	Probabilistic	Possibilistic
Basis	Probability theory	Possibility theory/ fuzzy logic
End-point type	Crisp events	Vague / fuzzy events
Exposure assessment	Probability density distribution	Possibility or necessity distribution
Effect assessment	Cumulative probability of effect conditional on exposure	Cumulative possibility/ necessity of effect conditional on exposure OR Implication operator OR rule-base
Likelihood characterisation	Product	Implication related t-norm/ t-conorm operator (e.g. min/ max)
Stressor likelihood integration	Sum-product rule	Max - min operators

2.7 DERIVING AND INFORMING STRESSOR RESPONSE RELATIONSHIPS

2.7.1 INTRODUCTION

In Figure 2.1 it is shown that the only parallel tasks in ERA are effect assessment and exposure assessment. Of these exposure assessment has the advantage of a number of models being available for predictive exposure assessment.. For substances, models such as WARNB, MCARLO and SIMCAT could be used with stochastic inputs to calculate effluent criteria and TOMCAT and QUAL2E in addition to the others can be use to estimate in-stream substance concentrations (Ragas, *et al.*, 1997). A number of flow models also exist (e.g. the Pitman model commonly used in South Africa (Pitman, 1973)). At present it is not known whether any models exist to predict habitat degradation as a stressor and it appears likely that habitat degradation will remain to be assessed *in situ*. Therefore, a combination of observation and modelling can be used to estimate the stressor exposure likelihood.

The other component in the risk estimate, effect likelihood, was characterised as the likelihood of effect conditional on the exposure as represented by the stressor-response-relationship (SRR). In its simplest form an SRR could be characterised by a lower and an upper acceptability limit as illustrated in Figure 2.9.

The **minimum characteristics** of an SRR for effect likelihood are:

1. It must express the relationship between a level of stressor and the level of occurrence of the end-point.

2. It should be able to resolve the stressor-levels where there is no expectation of end-point response and complete expectation of end-point response.

3. In its simplest form it could be a discontinuous stepped function as shown in Figure 2.9, but it could also be a smooth s-shaped curve. The form shown in Figure 2.9 indicates an increasing expectation as the stressor metric increases. The acceptance limits need not represent discontinuities but may be interpreted as selected percentiles of a suitable cumulative distribution curve or some other suitable function as long as it reflects the **present state of knowledge**. A SRR could also be in the form of a rule base.

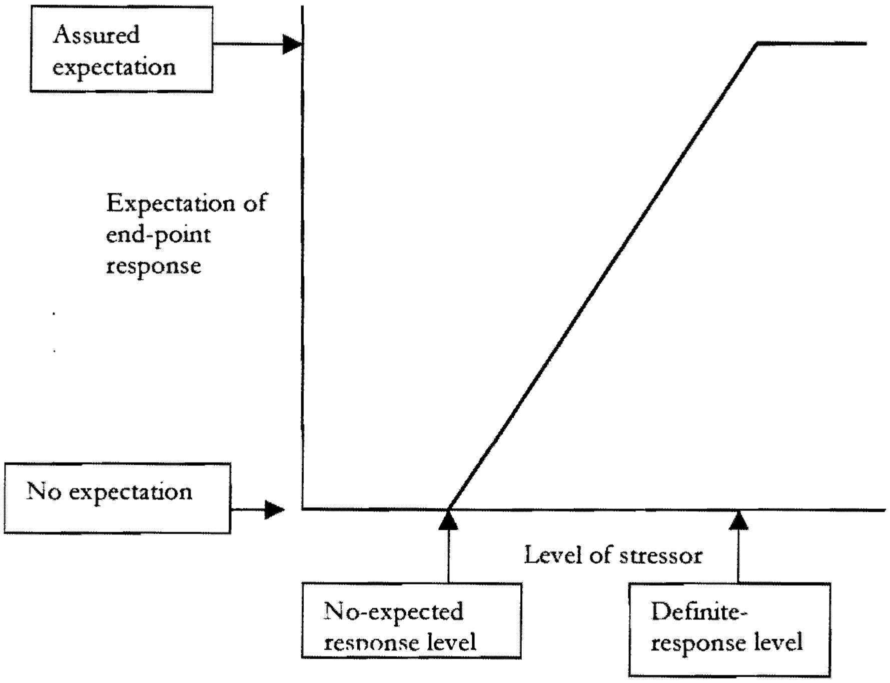

Figure 2.9 An illustration of the parameters needed to construct an SRR. The upper and lower expectation limits are stressor levels corresponding to unacceptable and acceptable levels of expectation of effect.

4. It must be **monotonic**, although it need not be strictly monotonic. That is, any given effect expectation should map to only one point or contiguous interval in the stressor level domain. A stressor that has a similar effect at very high and at very low levels should be modelled as two separate stressors. The reason for the monotonicity is to preserve consonance between the effect and stressor.

INFORMING TOXIC SUBSTANCE SRR'S

Ecotoxicological methods used in the bio-assessment of toxic substances are a solid basis for developing SRR's. The data as derived from toxicity tests serve as the basis for hazard assessment. Two common assumptions when applying these data in hazard assessment are (a) that exposures are temporally invariant and (b) that individual based tests apply directly at higher levels of organisation.

Effect data for toxic substances exist mostly at the *individual organism* level and, to a lesser extent, at the population level, while effect data for the other stressors exist largely t the population and community level. However, more realistic risk assessment is still hampered by a lack of knowledge of conditional probability of effect at higher levels of organisation. As a simplification, it is often assumed that an impact at the lower level of organisation (where the data exist) necessarily implies an impact at the higher level of organisation. Consequently, the risk predicted at the lower level of organisation is at least as great as that predicted at the higher level of organisation since the probability of a logical consequent cannot be greater than that of the antecedent. Although this is a reasonable starting point, if all the interactions have not been accounted for and the conditional probabilities evaluated, this assumption could be seriously in error. As a result, the calculation above, and indeed any risk assessment based on such a premise, could be seriously in error.

The assessment of the parameters in the temporally invariant case derives directly from ecotoxicological assessment. The higher the level of organisation represented in the test the better. Some notes on the use of population level projections from individual level assessments are made in 2.4.3. Temporally variable stressor levels are more realistically found in real stream quality management situations and these present a greater challenge. Some notes are appended on the estimation of probable mortality from temporally variable concentrations.

A brief overview of some of the issues involved in toxicity bio-assessment as the basis for toxicity SRR's appear in the Appendix 2.11. From this discussion it is clear that:

1. Since it is impossible to define a "most sensitive species" the estimation of a protection level is based statistical models. This implies the selection of toxicity test species should be as extensive as possible so that a suitable database can be generated for the statistical models.

2. While the bulk of toxicity data is generated at the individual organism level, this is generally not the best level for data on which to base ecosystem-level decisions. However, methods do exist to project from the individual organism-level to higher levels. These would include the methods referred to in Table 2.5.

3. Its is of particular importance to incorporate the effect of time variable toxic substance levels. Data on bioconcentration could be used very effectively in combination with pharmacokinetic models to estimate response expectation.

4. The interpretation and application of mixture toxicity data needs to be developed further in order to improve SRR quality.

Table 2.5 Some common methods for extrapolation of effects

Type	Extrapolation/ Projection	Form	Rationale	Reference
Bio-assessments	Stressor magnitude (e.g. concentration) to species level effect.	Concentration-response functions	Concentration proportional to receptor dose	e.g. Hathway, 1984
Response regression	From lower to next higher taxonomic level	Regression equations, projection matrices	Species representative of its taxon	Suter , 1993; Caswell, 1989; Suter, 1993; Caswell, 1996
Dose scaling	Across species	Allometric equations	Physiological functions proportional to physical characteristics (e.g. body mass, volume etc.)	Kenaga, 1978; Crouch, 1983; Chappell, 1992; Suter, 1993
Diet extrapolation	Across different trophic groups	Qualitative categories of susceptibility	Adaptation to common diet	Mullin, *et al.*, 1982; Suter, 1993
Guild extrapolation	Across different guilds	Qualitative similarities	Common diet and environment and similar behaviour within guilds	Cummins, 1974; Severinghaus, 1981

INFORMING FLOW AND HABITAT STRESSOR SRR'S

In contrast to toxic SRR's, the SRR's for flow and habitat stress is more likely to be derived from field observations with interpretation by experts in the field.

However, much work is being done from which flow-related stress and flow-related stressor-response information can be drawn (e.g. King and Louw, 1998; Hughes and Münster, 1999) and some experimental and or observational data exist from which the possibility of effect can be inferred (e.g. Chessman, *et al.*, 1987; Quinn, *et al.*, 1992; Cooper, 1993; Roux and Thirion, 1993; Thirion, 1993). It appears that much more research is needed to assess effects at *ecosystem* level.

An important feature of risk-based management is the feedback loop between the field bio-monitoring and the problem formulation and risk characterisation steps in risk assessment. Risk in itself cannot be proved to be correct or incorrect, but a formal methodology to adapt the process, will ensure dynamic, scientifically defensible risk management in a catchment.

From the discussion of an approach to derive habitat and flow SRR's it is clear that:

1. There is a dearth of information on habitat and flow stress and there is nowhere near the amount of controlled experimental data on which to base the SRR's compared to toxic SRR's. The use of a fuzzy expert system may in many cases be the only type of SRR available.

2. A fuzzy relationship of the form $E = \mathrm{R}\ o\ A$ may be used, where E is an effect, o is a suitable implication operator and A is a stimulus. R is the SRR for the stressor and would likely be in the form of a matrix.

3. In order to formulate R, there must exist a training set of stimuli and responses. Once R has been formulated it is applied in conjunction with observed or predicted stimuli to predict response expectation.

4. In the case of flow and habitat response, it is particularly necessary to develop the methodology to update R by using data from field observations. This can be done by the use of the Dempster-Schafer theory (DuBois and Prade, 1988). A considerable volume of work has been done on belief functions and their updating by Dempster-Schafer as well as other updating algorithms (Smets, 1981; 1991a,b; 1993; 1994).

2.9 CONCLUSIONS

The two major problems in applying risk methodology in ERBM relates to the effect assessment phase. This phase requires the formulation of a SRR, which must express the relationship between the stressor level and the expectation of the end-point effect. With regard to SRR's the two most obvious problems are: (a) the problem of estimating the risk at higher level end-points when only data at lower level end-points are available because the end-points are incompatible, and (b) informing the SRR.

The theoretical considerations presented in this chapter indicates that:

- Both uncertainty and variability are likely to be important in ERA and ERBM. There is clearly a need to ensure that risks are assessed at the correct organisational level and consequently there is a need to project the risk estimated at a lower organisational level to a higher organisational level. The uncertainty around end-point projection can be addressed by and phenomenological end-point projection model.
- The likelihood of ecological effect can be expressed either in probabilistic or possibilistic terms. The interpretations are compared in Table 2.4.
- A comparison of the form of Eqs. [2.12] and [2.15] shows that the probabilistic formulation will most likely yield the lower limit of expectation of the end-point while the possibilistic formulation will most likely yield the upper limit of expectation. Which one of the two is used will depend on the purpose of the risk assessment.

- Methods do exist to inform SRR's. Toxic SRR's can be based on the toxicity assessments. In this case it is particularly necessary that the risk end-points need to checked carefully. Other stressors, such as flow and habitat degradation, would more likely benefit from fuzzy expert system formulation of the SRR problem. In all cases, but especially in the case of flow and habitat stress, is it necessary to update the SRR from field observations. The challenge to risk management of multiple stressors will be the formulation of expert systems that are able to tap the ecological knowledge of the effect of stressors at higher levels of ecological organisation and express it in a form that can be used in ecological effect assessments. The assessment of the likelihood terms in the model is not a simple task.

- The choice of basis on which ecological effect likelihood is based should correspond to the characteristics of the end-point and nature of the data available. For crisp, well-defined events, which are uncertain in occurrence, a probabilistic formulation is well suited. If the end-point or the data is subject to epistemic uncertainty, then fuzzy logic and a possibilistic formulation is indicated.

CHAPTER 3

MODELLING THE DIVERSE STRESSOR PROBLEM

All models are wrong, but some are useful – George Box (1979)

3.1 SUMMARY

This Chapter deals with the problem of estimating the aggregate risk of a number of diverse stressors (referred to as the Diverse Stressor Problem).

There did not appear to be any formal mathematical formulation of ERA except for the Kelly-Roy-Harrison formulation. This formulation could be shown to be a special case of the probabilistic conjunction of stressor effect and stressor occurrence.

The aggregate risk of diverse stressors is modelled as the disjunctive occurrence of effects due to the different stressors. Both probabilistic and possibilistic formulations of this model were made and tested in hypothetical cases. These tests showed that the probabilistic formulation had more strenuous requirements regarding end-point definition and SRR input compared to the possibilistic formulation, but it is more likely to be applicable in law-enforcement. The fuzzy (possibilistic) formulation was more easily adapted to imprecise ecological data.

3.2 ESTIMATING THE AGGREGATE RISK OF DIVERSE STRESSORS: THE DIVERSE STRESSOR PROBLEM

3.2.1 THE KELLY-ROY-HARRISON EXPRESSION

Although the use of conditional probability (and other expressions of likelihood) is well known in risk assessment generally, it has not been obvious in literature on ERA. Kelly and Roy-Harrison (1998) note that mathematical formulation of ERA appears to be pointedly avoided for fear of misuse or misinterpretation. Nevertheless, they formulate risk (R) as a function of an adverse effect (E), the consequence of an adverse effect (C(E)) and the likelihood of adverse effect (Λ(E))which is expressed as a function of exposure (P) and the existence of a stressor (S) such that for *k* severity levels, *i* stressor levels and *j* exposure levels:

$$R = \sum_k C_k(E_k) \cdot \sum_i \sum_j \Lambda(E_k \mid P_{ji} \wedge S_i) \cdot \Lambda(P_{ji} \mid S_i) \cdot \Lambda(S_i) \qquad [3.1]$$

With regard to the Kelly-Roy-Harrison formulation (Eq. [3.1]) it should be noted that:

1. It makes provision for the situation where a stressor is given while the various consequences needs to be explored and quantified. In this study the focus is on the situation where the end-point is given (encapsulated in an ecosystem level phenomenon, e.g. loss of sustainability). This means that the consequences are discounted in the end-point and all that is left to determine is the likelihood of adverse effect. Furthermore, because ERBM focuses on management for a predetermined effect and its probability, both 'consequences' and 'adverse effect' (i.e. C(E)) is fixed by the regulatory requirements. Consequently, Eq. [3.1] practically reduces to Eq. [3.2].

$$R = \sum_k \sum_i \sum_j \Lambda(E_k \mid P_{ji} \wedge S_i) \cdot \Lambda(P_{ji} \mid S_i) \cdot \Lambda(S_i) \qquad [3.2]$$

2. Eq. [3.2] makes a distinction between stressor occurrence and exposure. In environmental assessment of the effect of chemicals, this is fundamentally correct because a stressor introduced into the environment may contact an organism by various routes simultaneously with each route contributing differently to the overall risk. In aquatic environments there may probably fewer routes of exposure and some are more likely to dominate. **In the short term**, direct intake of water is likely to dominate, while on the longer term indirect exposure may also contribute. In the view of Kelly and Roy-Harrison (*op. cit.*), for human and ecological risk assessment, Λ(S)=1. In other words, the stressor definitely occurs, it is only the exposure that may differ. For the purpose of this study, where for some stressors effect does not depend on uptake but on overall stress, it is assumed that occurrence and exposure are equivalent. It should be borne in mind that for chemicals (and particularly toxics) this assumption does not necessarily hold. For the purpose of this study Eq. [3.2] reduces to [3.3].

$$R = \sum_k \sum_i \Lambda(E_k \mid S_i) \cdot \Lambda(S_i) \qquad [3.3]$$

3. Eq. [3.3] still contains the summation over *k* severity levels of adverse effect and *i* stressor levels. Probability is expressed as probability density and consequently Eq. [3.3] is an expression of the area overlap between effect and exposure distributions. This stands in stark contrast to the calculation of risk by the quotient method (See Risk Characterisation Phase in Section 2.2.2) where two concentrations or stressor levels are compared. Eq. [3.3] is a more general form of risk expression.

4. Eq.[3.3] contains an expression of a SRR and a stressor occurrence expectation. This is in fact a special case of the probabilistic expression of the ***modus ponens*** inference (Eq. [2.1]) where the rule $S_i \rightarrow E_k$ and the observation S_i are combined (i.e. $R = \Lambda(S_i \rightarrow E_k \wedge S_i)$). This expression is analogous to the combination of the inferences in Eq. [2.9]. A more general expression that does not prescribe the way in which likelihood is to be expressed needs to be derived.

The other major problem still remains: how to estimate the aggregate risk when a number of different stressors occur.

3.2.2 CONJUNCTION-DISJUNCTION EXPRESSION

From the theoretical considerations in Chapter 2 it was established that a risk only occurs when (a) a stressor exists AND (b) the stressor (by definition) has an effect on some target entity in the ecosystem. Therefore, if the stressor existence is designated by S and the effect of the stressor is designated by E then a risk only exists when $(E \wedge S)$ is true. More precisely the risk is the likelihood that $(E \wedge S)$ is true: $R = \Lambda(E \wedge S)$.

The effect E is here a generalised expression of the observation that a stressor of the same type as S has an effect. This effect generally occurs over stressor set Y. However, risk is assessed for a specific situation, where particular values of Y, namely the set S will be found (i.e. $S \in X$). So risk for stressor X is the properly expressed as Eq. [3.4]

$$R_X = \Lambda((E_X \mid X) \wedge S) \qquad [3.4]$$

If likelihood is expressed in terms of probability then Eq. [3.4] becomes Eq. [3.5] while if it is expressed as possibility then it becomes Eq. [3.6]

$$R_X = P((E_X \mid X) \wedge S) \qquad [3.5]$$
$$R_X = \Pi((E_X \mid X) \wedge S) \qquad [3.6]$$

The effect E could, in the present context, be the occurrence of an event such as "loss of sustainability". Each stressor acting on an ecosystem may result in E either on its own or in conjunction with other stressors. So each stressor produces an individual risk of effect E. If stressors X, Y, Z… are present in the system and they occur on a site-specific basis as S, T, U, …, then the risk R of E due to either X OR Y OR Z OR … will be given by Eq.[3.7].

$$R = \Lambda\{((E_X \mid X) \wedge S) \vee ((E_Y \mid Y) \wedge T) \vee ((E_Z \mid Z) \wedge U) \vee \ldots\} \qquad [3.7]$$

R is an expression of the aggregate risk and is assessed in a manner similar to Eqs. [3.5] or [3.6]. Each of these individual stressor risks can be estimated by ERA. In order to assess the expectation of all the stressors acting at the same time, the individual stressor ERA outcomes need to be convoluted. There are several mathematical operators that can be used to convolute stressor risk to reflect the total risk, including: maximum, sum and conjunction. The specific operators will depend on whether a probabilistic or possibilistic formulation is used. These will be investigated in section 3.3 and 3.4 respectively. The event *E* will, in the rest of the Chapter, be partitioned into events that relate to the various types of anthropogenic stress, such as toxicity (*t*), flow regime disturbances (*q*) and habitat degradation (*h*).

3.3 PROBABILISTIC AGGREGATE OF DIVERSE STRESSOR RISK

3.3.1 BACKGROUND

In a probabilistic expression of the aggregate risk consider the event *E* in an ecosystem subject to *n* different stressors. Each stressor *i* will give rise to E_i. The combined probability of effect (in set theoretical terms) is given by (DeFinetti, 1990):

$$P(E) = P\left(\bigcup_{i=1}^{n} E_i\right) = \sum_i P(E_i) - \sum_{i,j} P(E_i E_j) + \sum_{i,j,h} P(E_i E_j E_h) - \ldots \pm P(E_1 E_2 \ldots E_n) \qquad [3.8]$$

If E_t , E_q and E_h are all logically independent, then probability of the conjunction of individual ecological effects reduces to the product of the individual effect probabilities, and hence the application of Eq. [3.8] to Eq. [3.7] yields Eq. [3.9]:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)] \qquad [3.9]$$

3.3.2 SYNERGISM OR ANATAGONISM AMONG STRESSORS

$P(\varepsilon_x | x)$ is defined as the probability of an end-point ε given the event that stressor X is present at level *x*. Furthermore, the effects ε_x may not be functions of one stressor only. It may be necessary to partition the event "existence of stressor *X*" into events that signify the occurrence of stressors that collectively manifest as stressor *X*: i.e. *X* is partitioned into occurrence of stressors (X_1 , X_2 , ...X_n), where there are *n* stressors that make up the class of stressor *X*. Due to interactions among stressors, it may be necessary to evaluate $P(\varepsilon_x|X)$ where all *n* different stressors are present at the same time. Most often this will not be possible experimentally (except perhaps in the case of toxic stress), so that simplifying assumptions will have to be made. However if stressor occurrence events X_j are logically independent then this reduces to Eq. [3.10] (DeFinetti, 1990).

$$P(\varepsilon_x|X) = \sum_j \left(P(X_j) \cdot P(\varepsilon_x|X_j)\right) \qquad [3.10]$$

It might be, that although the stressor occurrences X_i and X_j are independent, the effect ε is dependent on the co-occurrence of X_i and X_j . This might be due to some mechanistic interdependence such as synergism or antagonism in which case the occurrence of (X_i,X_j) might manifest as a new stressor Y. In this case $P(\varepsilon|X_i X_j)$ would be given by: $P(\varepsilon_Y|Y) = P(\varepsilon,Y)/P(Y)$. Therefore, $P(\varepsilon,X_i,X_j) = P(X_i)P(X_j)P(\varepsilon|Y)$, where the value for $P(\varepsilon|Y)$ has to be evaluated experimentally. However, cases of true synergism among toxics, for example, are reported to be rare (Calamari and Vighi, 1992). The occurrence of synergism among other stressors may be possible.

3.3.3 A HYPOTHETICAL CASE STUDY

A hypothetical case study to illustrate an application of the above is given in Part 3, Paper 2.

3.4 POSSIBILISTIC AGGREGATE OF DIVERSE STRESSOR RISK

3.4.1 THEORETICAL BACKGROUND

The point of departure in formulating of aggregate ecological risk is Eq. [3.7]. Rewriting Eq. [3.7] for the three-stressor assumption yields Eq. [3.11]

$$R = \Lambda\{((E_Q\,|Q) \wedge q) \vee ((E_H\,|H) \wedge h) \vee ((E_T\,|T) \wedge t)\} \qquad [3.11]$$

The possibilistic approach to the ecological risk problem is formulated as the disjunction of the ecological risk rule base with predicted or observed stressor data. The risk rule is captured in the conditional likelihood. E is defined by the NWA as being "loss of sustainability" or $\neg Sus$. Each of the disjunctive terms in t Eq. [3.11] can be written in the form:

$$\begin{array}{lll} \text{Rule:} & \text{X is A} \rightarrow \neg\,\text{SusY is B} & \\ \text{Observation:} & \underline{\text{X is A}'\qquad\qquad\qquad}. & [3.12] \\ \text{Conclusion:} & \qquad\qquad \neg\text{Sus is B}' & \end{array}$$

Each premise contains a characteristic ("sustainability") and an evaluation ("loss of"). In the case where the propositions in the premise can only be true or false (i.e. the application of "crisp" logic), the uncertainty is expressed in terms of probabilities.

The evaluation of the propositions in the case of most ecosystems is almost necessarily vague, epistemic of an observer in a situation and possibly phenomenological. In general, probabilities cannot be used to evaluate the likelihood of effect. In order to apply the well-established probability calculus to the estimation, the evaluations are given a numeric value so that Aristotelian logic applies. For example, if the evaluation "maintained" is replaced by "80% maintained" then the outcome of an assessment can be true or false in principle. This, however, requires either considerable ecosystem specific knowledge, or, simply assumption of a value as a norm. The nature of ecological assessments is often more amenable to vague assessments of these values such as: "high", "moderate" etc., which corresponds to typical fuzzy sets. So, the expressions A and B in [3.12] are fuzzy sets. Consequently, if t is a specific response to stimulus s, then Eq. [3.12] can be solved by (DuBois and Prade, 1988):

$$\mu_{B'}(t) = \sup_{s \in S}(\mu_A(s) {*}{\rightarrow}\, \mu_B(t)) * \mu_{A'}(s) \qquad [3.13]$$

where * is a suitable t-norm and *→ is the corresponding implication operator which could be replaced by the conditional possibility distribution $\pi_{Y|X}(s,t)$ if the sets are normalised.

In this study the evaluation was performed for four fuzzy sets so that A, B ∈ { Negligible, Low, Moderate, High}. For example [3.11] can be expressed as "IF effect of stressor 1 IS Negligible OR effect of stressor 2 IS Negligible OR… THEN NOT (Sustainability) IS Negligible"

For each stressor, $Poss(E_i)$ and $Nec(E_i)$ can be calculated (DuBois and Prade, 1988; Kruse, *et al.*, 1994):

$$Poss(E_1 \vee E_2 \vee E_3 \ldots) = \max\{Poss(E_1), Poss(E_2), Poss(E_3)\ldots\} \text{ and}$$
$$Nec(E_1 \vee E_2 \vee E_3 \ldots) \geq \max\{Nec(E_1), Nec(E_2), Nec(E_3) \ldots\} \qquad [3.14]$$

A more complete expression of the risk inference in terms of a conditional possibility or necessity measure (DuBois and Prade, 1988) is:

$$\begin{aligned} &Poss\,(X \mid E_X) \geq a' \\ &Poss\,(E_X \mid X) \geq a \qquad [3.15] \\ &\underline{Poss\,(X) \in [b, b']} \\ &Poss\,(E_X) \in [a{*}b, a'{*}{\rightarrow}b'] \end{aligned}$$

$$\begin{aligned} &Nec(X \mid E_X) \geq a \\ &Nec(E_X \mid X) \geq a' \qquad [3.16] \\ &Nec(X) \in [b, b'] \\ &Nec(E_X) \in [\min(a, b), (1 \text{ if } a' \leq b' \text{ or } b' \text{ if } a' > b')] \end{aligned}$$

The possibility and necessity measure are interpreted to mean the extent to which a fuzzy set may possibly correspond to a given description and the extent to which a fuzzy set may correspond to the complement of the fuzzy set respectively. For the probability measure, P, of set E_X, it is always true that $Nec(E_X) \leq P(E_X) \leq Poss(E_X)$. Consequently, it is possible to estimate the upper and lower limits for the possibilistic risk to the ecological sustainability from a knowledge of the possibility and necessity of the stressor levels which can be calculated from the possibility distributions of the stressors, the stressor response and some knowledge of the stressor impact structural biodiversity inference.

3.4.2 HYPOTHETICAL CASE STUDY

A hypothetical case study is described in Part 3, Paper 3.

3.5 INDEPENDENCE OF PHENOMENA

In the foregoing, the assumption of independence of phenomena featured strongly. One of the strongest objections to Jooste (2000) had been the assumption of independence among stressor phenomena. It was pointed out that it is well known that some substances act synergistically even though true synergism is reportedly quite rare. Furthermore, even among heterogeneous stressors it is quite conceivable that when two stressors occur together (e.g. flow insufficiency and toxic substances) that the stress caused by the one exacerbates the stress caused by the other, and although there is no true synergism, the effect would be qualitatively similar.

This objection appears to be due to the "Mind Projection Fallacy" (Jaynes, 1996) at work in risk assessment. It should be remembered that risk, although often expressed as a probability, is in fact a descriptor of the assessor's *state of knowledge*, assigned to a *phenomenon*. While it may incorporate knowledge of the mechanistic detail, once the descriptor for a particular set of stressor values is assigned, it loses that detail.

Consider a multiple stressor problem as follows: Assume that the phenomenon: {Unsustainability is caused by stressor **x** with value x } is indicated by X. Assume that stressor **y** with value y resulting in stress Y occurs simultaneously. It is important to note that a distinction is made between the *phenomenon* and the *mechanism* by which this phenomenon came about. For the risk assessment of X it would be important to know by which different mechanisms the phenomenon X was reached. If , for example, a probabilistic risk of X is considered then the risk would be given by $P(X|x \wedge y)$. This can be recognised as a Bayesian posterior distribution, which is the left-hand side of Eq. [3.17].

$$P(X \mid x \wedge y) = P(X \mid x) \cdot \frac{P(y \mid X \wedge x)}{P(y \mid x)} \qquad [3.17]$$

In general, the question should be asked in risk assessment whether there exists any knowledge of the likelihood ratio (i.e. the second term on the right hand side of the Bayes equation). The prior probability must, by definition, exist since that is the rationale for doing a risk assessment.

An assessment of the likelihood ratio begs the question of whether the existence of stressor value could have been inferred from a knowledge of the existence of stress X and the co-occurrence of stressor values x and y. In general it might be suspected that such a synergism exists, but proof is often lacking. If there is mechanistic reason to believe that y will potentiate (or exacerbate) the effect of x, then an assessment of the likelihood ratio can in principle be done. If no evidence exists, then the posterior probability equals the prior probability and the risk pertaining to the co-occurrence of the two stressors is no different from the risk of induced by x, i.e. the likelihood ratio is 1. However, if the likelihood ratio differs from 1 then the risk pertaining to the phenomenon X is given by the posterior distribution. The stressor values and their interaction have now been discounted in the risk calculation. Consequently, the risk of X for any given set of x and y will be independent of risk of Y. Therefore, it could be said that the risk of the phenomena X and Y are logically independent. So, although some causal dependence may exist, the risk of the phenomena may be logically independent. It seems particularly prudent in ecological risk assessment to be wary of the "Mind Projection Fallacy" (see below)

Jaynes (1996 *p* 406) describes the difference between causal and logical independence as follows: "Two events may in fact be causally dependent (i.e. one influences the other); but for the scientist who has not yet discovered this, the probabilities representing his state of knowledge – which determine the only inference he is able to make – might be independent. On the other hand, two events might be causally independent in the sense that neither exerts any causal influence on the other [...] yet we perceive a logical connection between them, so that new information about the one changes our state of knowledge of the other. Then for us their probabilities are not independent." He described this confusion between reality and a state of knowledge about reality as the "Mind Projection Fallacy".

3.6 AGGREGATION MODEL SUMMARY

The aggregation of the risk of diverse stressors make of the logical disjunction of individual stressor risk.

$$R = \Lambda\{((E_X \mid X) \wedge S) \vee ((E_Y \mid Y) \wedge T) \vee ((E_Z \mid Z) \wedge U) \vee \ldots\}$$

In probabilistic terms this model becomes:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)]$$

In possibilistic terms this model becomes:

$$Poss(E_t \vee E_q \vee E_h) = \max\{Poss(E_t), Poss(E_q), Poss(E_h)\} \text{ and}$$

$$Nec(E_t \vee E_q \vee E_h) \geq \max\{Nec(E_t), Nec(E_q), Nec(E_h)\}$$

The individual stressor risks are calculated from a SRR and a likelihood of stressor occurrence.

In probabilistic terms:

$$P(E_X) = P(\varepsilon_x \mid x) \cdot P(x)$$

In possibilistic terms:

For a fuzzy descriptive set A or A′of stressor X and fuzzy descriptive set B or B′ of response Y:

$$\mu_{B'}(t) = \sup_{s \in S}(\mu_A(s)^* \rightarrow \mu_B(t)) * \mu_{A'}(s)$$

Poss (B′) = max { $\mu B'_i(t_i)$} over all stressors i

A comparison between the probabilistic and possibilistic formulation in Table 3.5 below shows that, at least in the short term, the fuzzy formulation might be more appropriate, although the regulatory requirement might motivate for clarifying the knowledge-base to allow for the use of the probabilistic formulation.

Table 3.5. A comparison between the probabilistic and possibilistic formulations of the diverse stressor problem.

Component	Probabilistic	Possiblistic
End-point	Crisp definition	Fuzzy or crisp definition
SRR-type	Unique	Unique or fuzzy
SRR data requirement	Extensive	Limited, expert system
Adaptability to diverse ecological stressors	Low (data limitations)	High
Applicability of results to law-enforcement	Well adapted	Difficult

CHAPTER 4

MODELLING THE DIVERSE-STRESSOR-MULTIPLE-SOURCE PROBLEM

4.1 SUMMARY

This Chapter deals with the solution to the diverse-stressor-multiple-source (DSMS) problem in the context of ecological risk-based management (ERBM). The effect disjunction solution to the diverse-source problem of Chapter 3 is used as the basis for solving the DSMS problem. ERBM requires that stressor profiles be generated from risk objectives. This is accomplished by setting the risk objectives equal to aggregate risk in the disjunctive formulation. The stressor profiles may best be generated either by setting risk-based in-stream stressor objectives (which requires a waste load allocation to generate source-specific criteria) or by iterative selection of stressor profiles and comparison of the aggregate risk to the objective. The most flexible, but computationally the most intensive solution is the iterative selection of stressor profiles.

In order to select among the infinite number of solutions, the DSMS problem is formulated as an optimisation problem that seeks to find the stressor values based on the maximum degree of acceptability of the outcome to all role players. It is proposed that regulatory satisfaction will be determined by satisfaction of the risk objective while stressor sources' satisfaction will be determined by the degree to which the stressor reduction requirement will impact on technical, economic or other issues. The overall degree of satisfaction, λ, is made up of the regulatory risk satisfaction λ_R as well as λ_{ij}, the source i, stressor j specific degree of satisfaction. The calculation of λ both as the average over all λ_{ij} and the minimum over all λ_{ij} were investigated.

The control variable was chosen as the fraction of the "raw" stressor that is allowed from the source (i.e. the stressor attenuation), x. Besides the obvious constraint that $x \in [0, 1]$ the use of an equity constraint (which requires that all stressors of the same type be treated equally), and a minimum level for λ_R may also serve as constraints. The impact of each of these has been evaluated in a hypothetical test case.

A genetic algorithm appeared to be a more effective in solving the optimisation problem than the variable simplex. The genes were composed of the set of attenuation values. The initial population of 20 individuals was selected from the randomly generated attenuation values that satisfied the constraints. The individuals were ranked according to decreased λ. The next

generation was produced by sexual reproduction selecting the parents randomly from an exponential distribution and retaining the five best individuals with 15 child individuals. The new genes were generated by random crossover between parents with a mutation rate of 0.01. A published technique was used that focussed the control variable search domain after every 5 generations.

It was shown that despite the significant computation time (about 3hours for a case of 3 stressors and 4 sources on a 333MHz Pentium processor with QBASIC as coding language) satisfactory results could be obtained. From the optimal attenuation levels, source-specific stressor management criteria can be generated.

4.2 ASPECTS OF THE ECOLOGICAL RISK MANAGEMENT PROBLEM

4.2.1 BACKGROUND

Water resource management in the context of the protection of the aquatic ecosystem subject to multiple sources of a variety of stressors has to deal with:

1) the problem of setting goal-related management objectives for substantively diverse stressors and
2) an equitable and transparent apportionment of the impact among the users of the resource.

The risk assessment problem, where the risk pertains to a given combination of stressors, has to answer the question: "What is the likelihood of effect if the given stressors occur with a given likelihood?" In general the water resource risk management problem has to answer the question: "What should the stressor levels be (or stressor distribution be) if an in-stream risk target needs to be reached?" In the latter case a risk level is set and the goal and management objectives need to be derived which comply with that goal.

4.2.2 OPTIONS IN SOLVING THE DSMS PROBLEM

The diverse stressor model that had been developed in the previous chapter addressed the risk assessment question. It was shown that for ERA the conjunctive convolution of individual stressor risk could reasonably be used to estimate the aggregate risk. For stressors X all resulting in a specific unacceptable effect ($X \in \{T, Q, H\}$) the aggregate risk will be given by either version of Eq. [4.1].

$$R = \Lambda\{((E_T \mid T) \wedge T) \vee ((E_Q \mid Q) \wedge Q) \vee ((E_H \mid H) \wedge H)\}$$

With the assumption of independence this yields:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)]$$

or: [4.1]

$$Poss(E_t \vee E_q \vee E_h) = \max\{Poss(E_t), Poss(E_q), Poss(E_h))$$

Each of these individual stressor risks is calculated from an equation of the form: $\Lambda(E_x) = \Lambda\{(\varepsilon \mid x) \wedge x\}$ where Λ is a measure of likelihood like probability or possibility. Therefore the risk is a conjunction of a SRR and a stressor occurrence.

There may now be three approaches to answer the risk management question:

a) a deconvolution of aggregate stressor risk into individual stressor risk, or

b) setting stressor-specific risk-based instream objectives., or

c) an iterative solution of the risk assessment question based on selected stressor values stopping when the aggregate risk equals the target risk (within selected precision bounds)

a) DECONVOLUTION

The deconvolution option, which seems at first appears to be the most attractive, is shown on reflection to be almost intractable. Each of the individual stressor risk terms is itself the product of two uncertain and/or variable terms, one derived from the stressor response relationship and the other from the stressor exposure. The deconvolution would therefore have to be performed in two dimensions, which decreases the tractability.

b) RISK-BASED INSTREAM OBJECTIVES

In ERA, both the SRR and the stressor can be subject to variability and uncertainty. The uncertainty in the SRR can be addressed by reducing this relationship through the assumption of a level of effect that represents in some way a minimally acceptable adverse effect. This would be analogous to using values such as the SAWQG criteria (Roux, *et al.*, 1996) except the SAWQG criteria are hazard-based rather than risk-based. In-stream stressor specific objectives, such as the South African Water Quality Guidelines (DWAF, 1996) may well reflect the regulatory goal, but is does not directly address the end-of-pipe or point-of-introduction criterion that is of importance to both the law enforcement agency and the user (discharger or abstractor). In its simplest form the quality criteria set at an in-stream point can be translated to end-of pipe values by a waste load allocation (WLA). A number of models have been used in order to accomplish this, varying from simple deterministic dilution models to stochastic dynamic models incorporating various kinetic effects (Lohani and Thanh, 1987, Chadderton and Miller, 1981, Chadderton and Kropp, 1985, Tung, 1992, Cardwell and Ellis, 1993). In principle the same may be true for water quantity or any other ecosystem stressor.

Assimilative capacity

The normal practice of waste load allocations assumes that an "assimilative capacity" exists within a receiving water body (Foran and Fink, 1993). The assimilative capacity depends on the existence of an acceptable stressor level (ASL) as a management objective corresponding to an acceptable effect level (AEL), which relates to a management goal. The capacity of the system to function "normally" in the presence of the stressor is defined as the difference between the background or natural stressor level and the ASL. This stressor "capacity" is then "allocated" among sources of the stressor.

It should be recognised that the ASL is based on assumption and its validity is therefore dependent on the validity of the assumption. Even where a natural physiological threshold exists for individual response, the natural variability within populations and between communities in ecosystems causes thresholds to uncertain quantities. Consequently, ASL is naturally uncertain and strictly only stochastic WLA methods are valid.

Problems in using generic effluent criteria

To determine what level of stressor should be allowed at the point where the stressor is induced into the system requires a set of generic effluent quality criteria (such as the "general standard" that had been applied in South Africa for a number of years (DWAF, 1986)). However, such generic stressor specific criteria, while administratively useful, do not explicitly recognise:

- The uncertainty and vagueness often inherent in ecosystem knowledge and which is dependent on expert input. Numerical management criteria are created by the projection of a set of assumptions and (possibly) value judgements onto scientific data to reduce the impact of uncertainty, creating artificial discretisations in the situation assessment space. The resulting discontinuities in situation assessments, if not used circumspectly, lead to: a) unwarranted confidence in assessment results and b) reduces the system management flexibility. Not recognising the uncertainty, variability and possibly vagueness underlying the numeric stressor-specific criteria may lead to inappropriate allocation of resources to perceived rather than real problems and induces an unnecessary conflict potential into the management process.
- The contribution of diverse stressors to the same ecological phenomenon such as loss of sustainability. This leads to the anomalous situations: a) where all stressors may comply individually and yet the management goal is not attained (e.g. Dickens and Graham, 1998), or b) the system is managed assiduously for some perceived stressors while others are not considered at all, possibly because no management criteria exist for them.
- The specific needs of users and regulators that affect the acceptability of end-of-pipe criteria. The regulatory mandate to protect the aquatic ecosystem may be perceived to be in conflict with the economic and technological constraints of the discharges. Partially, this is the result of different paradigms in which the efficacy of criteria can be assessed. Management of a river system may pit an apparently ethereal value judgement of an ecosystem against the utilitarian demands by other water users.

- Not all dischargers can achieve any given level of treatment due to economic constraints. The source- and stressor-specific upper bound to the treatment level needs to be accommodated.

c) ITERATIVE RISK ASSESSMENT

The iterative solution uses the diverse stressor risk assessment formulation iteratively with a new selection of stressor values at each iteration. It then compares the aggregate risk calculated in this way to the risk objective.

Risk in the multiple source problem

Recognising the risk principle often underlying the derivation of stressor specific criteria, a flexible management tool for deriving stressor source attenuation criteria can be created by combining ecological risk concepts with WLA. This investigation starts with the premises that:

- some stress is inevitable when water resources are being utilised,
- there may be a specific situation where stressor-specific water resource objectives are insufficient to resolve conflicting interests and the extent to which stressors need to be attenuated needs to be negotiated,
- both regulator and users are able to formulate their criteria for acceptability (for the regulator in terms of risk and for the users in terms of the degree of attenuation), and
- enough expert knowledge and/or data exist to estimate the likelihood of a common ecological end-point for all relevant stressors.

Risk objectives

Once the WLA process is in operation, the sense of effect from which it originated, is lost. The process is inclined to consider the allocation of capacity independent of effect since the allocation is done in terms of stressor metrics. Replacing the hazard-based management objectives with risk (or effect-likelihood) objectives retains the sense of *effect* management as opposed to *stressor* management. The adoption of risk objectives would help to address these issues in terms of managing multiple sources of diverse stressors.

In the context of objectives, risk:

- is used here in the sense of an expression of the likelihood of observing a specified (unacceptable) effect as a result of a stressor (such as a toxic chemical) exposure (Bartell, *et al*, 1992) and therefore explicitly recognises variability and uncertainty (Suter, 1993),
- contains elements of likelihood, target and end-point (unacceptable effect):- all of which requires explicit statement

- is able to aggregate diverse stressors (see Part 2, Paper2 and Paper 3) through its expression in terms of likelihood, and with a suitable choice of end-point, is a dimensionless expression of expectation.

The actual value of the risk objective may be a matter of policy or negotiation.

Risk-based objectives would result in stressor specific criterion values, which are based on risk objectives, which are regulatory or societally expression of acceptability.

Discretisation of the risk continuum

The expectation of effect is assumed to have a monotonic relationship to the stressor level. This would imply that a point could be reached where the expectation is low enough to be of no further concern. This gives rise to the concept of a *de minimis* likelihood (or clearly trivial likelihood, from the legal term *de minimis non curat lex* – the law does not concern itself with trifles). Between the *de minimis* likelihood level and the *de manifestis* (or clearly unacceptable) likelihood level, there is a continuum of likelihood, which, for administrative purposes can be discretised into a series of acceptable levels of likelihood. Each of these risk objective values may itself be uncertain and only known by a clearly compliant value and a clearly non-compliant value.

4.3 FORMULATION OF THE DSMS-PROBLEM AS AN OPTIMISATION PROBLEM

4.3.1 BACKGROUND

The protection of a utilisable resource, such as water, may lead to a conflict of purpose between, on the one hand, the management agency charged with the protection of the resource and, on the other hand, the users intent on using the resource to the full. This management problem could be described in terms of a multiple objective optimisation among the conflicting goals of the role players (Sasikumar and Mujumdar, 1997). Although this is a simple problem in principle, the variability (stochasticity) and uncertainty inherent in the system and its management components are complicating factors that need at least a stochastic approach (Lohani and Thanh, 1978, Burn and McBean, 1985,Tung, 1992).

Optimisation refers to the process of finding the most favourable or best among a number of options. The solution to the diverse stressor problem proposed in the previous chapter made use of a disjunctive convolution of individual stressor risks as means of expressing the aggregate risk of the diverse stressors.

For any given value of aggregate risk, there are theoretically an infinite number of combinations of individual stressor risk levels that all result in the same aggregate risk. Each individual stressor risk level may in turn translate to an infinite number of stressor magnitude levels. If the risk-based approach to resource management is to be practical, the means need to exist to find the most favourable combination of stressor levels according to some relevant criterion.

The optimisation approach is well established in water resource management (Table 4.1)

Table 4.1 A review of optimisation techniques applied to water resource management. DO=Dissolved oxygen, BOD= Biochemical oxygen demand, COD=Chemical oxygen demand

Mathematical programming technique	Objective Function	Constraints	Special feature	Reference
Linear Programming (LP)	Cost minimisation	DO criteria	• River DO profile based in linear approximations of relevant differential equations • Mixed integer versions based on extended Streeter-Phelps model. • Parameters of the DO model, stream flow, waste flow and effluent BOD are stochastic parameters • Includes uncertainty in terms of design scenario's (see Notes)	Deininger, (1965) Loucks *et al.*, (1967) Lohani and Saleemi (1982). Hathorn and Tung (1989); Burn and Lence (1992)
Non-linear programming (NLP)	Cost minimisation Ditto	DO criteria DO criteria, seasonality of flow and treatment plant operation.	• Different river systems • Use of MINOS NLP software	Hwang, et al., (1973), Bayer, (1974). Herbay, et al., (1983).
Stochastic programming (SP)	Minimise cost	Stochastic BOD and COD	• Waste water treatment efficiency as variable	Ellis, (1987)
Dynamic programming (DP)	Minimise net cost Minimise DO deficit (Weighted objectives)	BOD constraints DO constraints	• Different waste water treatment options at each discharge point • Some use Monte Carlo simulation in water quality model	Dysart (1969), Futagami (1970), Newsome (1972), Hahn and Cembrowitz (1981), Joshi and Modak (1987).
Stochastic dynamic programming		Restrict or minimise number of standard violations Minimise magnitude of standard violation	• Use of sophisticated water quality models (WASP4 and QUAL2E) • Incorporates model (Type I) and parameter (Type II) uncertainty by regret modelling	Cardwell and Ellis (1993)
Chance constrained programming	Multi-objective: Treatment cost and water quality	Stochastic pollutant input	• Chance constraints	Boon, et al., (1989)
Fuzzy linear programming	Multi-objective (8 objectives including water quality and failure duration)	Evaluation criteria for objectives.	• Weighting of objectives • Uses fuzzy distance based ranking	Duckstein, et al., (1994)
Fuzzy chance constrained programming	Satisficing of operational risk objectives	Physical parameters of system operation	• Selection of a fuzzy risk level • Heuristic search algorithm for optimisation	Savic and Simonovic, (1991)

What is apparent in these optimisations is a) the preponderance of DO as a variable, b) the absence of ecological end-points in the problem formulation, and c) the absence of risk as a basis for optimisation.

The ecological implication of "DO deficit" is never explicitly addressed and is held as a vague and amorphous threat, which, if successfully removed, will result in some undefined benefit. The reason for the preponderance to DO modelling may be the result of two (possibly related) factors:

- The ubiquity of organic rich wastes from municipal and industrial waste-water treatment facilities, and
- The perception from legislation in many countries that oxygen depletion is the main cause of ecological stress in surface water.

While the latter may at times be a major factor determining ecosystem processes, it has also become increasingly clear that there are other stressors that are also important (See for example Dickens and Graham, (1998) and the literature cited therein).

There appears to be no alternative but to extend the optimisation process to include multiple stressors in order to solve the multiple-stressor-multiple-source problem. The optimisation problem formulation proceeds in four steps 1) formulating the philosophical point of departure, 2) isolating the pertinent stressors, 3) formulating the stressor occurrence and effect likelihood and 4) calculating the value of the objective function.

4.3.2 POINT OF DEPARTURE

It was assumed that:

1) South Africa, as an semi-arid, relatively poor country with a dependence on ecotourism would require that water resources be managed for maximum return flow, minimum stressor attenuation while striving to attain ecological protection goals. All of these requirements are of course not generally true, but it represents a precautionary scenario.
2) There exists enough goodwill and a spirit of co-operation between regulator and regulatees to solve the catchment management problem and for both parties to be willing to objectively formulate acceptability criteria in order to reach a compromise solution and that, above all, the regulatory framework allows for such a compromise.
3) The solution to the problem will be determined by the goal directed considerations informed by technology and economic considerations.

The implications of this point of departure is that:

a) All wastewater needs to be returned to a surface water resource. The National Water Act demands that no user may impair the sustainability of the water resource and, therefore, the contaminants in water that impact on the aquatic ecosystem need to be attenuated,

b) The best available technology from a Developed World point of view may not always be available to each stressor source and that homogeneous stressor attenuation levels may not always be feasible although it would the ideal,

c) Socio-economic or other "soft" (non-technical) factors may influence the extent and level of stressor attenuation and water resource protection (Beck, 1997). Each level of stressor attenuation carries with it an implication for the users and the ecosystem. These implications are likely to be interpreted in terms of diverse and possibly incompatible metrics. For example, the discharger may interpret a reduction of the allowable discharge of toxic substances in terms of treatment cost, employment opportunities lost as a result of inability to meet regulatory standards etc. On the other hand, the regulatory authority, charged with the protection of the aquatic ecosystem, interprets the attenuation level in terms of the threat to the long-term sustainability to the system. If the metrics of interpretation are not brought onto a common footing, the conflict may become irresolvable.

d) One source of communality between the user and the regulator is the acceptability of the regulated situation. The acceptability of different levels of stressor attenuation is likely to be epistemic so that it can best be described by a fuzzy set. This implies that acceptability can be graded in terms of degree of acceptability or conformity to the descriptor "acceptable".

e) The style of management on the part of the regulator would allow for explicit goal-oriented management and that these goals can be captured in risk values.

4.3.3 OBJECTIVE FUNCTION AND CONSTRAINTS

Generically, optimisation requires two components: an objective function expressing which values are to be minimised or maximised and (optionally) the constraints under which the optimisation should operate. The format of the problem would be:

> Maximise (minimise) the OBJECTIVE, which is a FUNCTION of CONTROL VARIABLE
>
> So that CONSTRAINTS are satisfied

OBJECTIVE FUNCTION

For the formulation of an objective function, communality between the regulator and the regulatee needs to be established.

- Under the NWA, the regulator is primarily concerned with the protection of aquatic ecosystem and this could be expressed in terms of the minimisation of ecological risk.

- The regulatee would have socio-economic and technical considerations as prime concern.

> The extent to which each role player is satisfied with the outcome of the regulatory process, is a common denominator in the sense of representing a common measure. This degree of satisfaction is designated by λ; the degree of satisfaction obtained with the level of risk achieved λ_R, while the degree of satisfaction of the manager of source *i* with the regard to stressor *j* is λ_{ij}.

The satisfaction of all regulatees can be aggregated into λ_x. The value of λ_x could be derived in two different ways:

> Option 1: The minimum acceptability over all controllable stressors at each source could be calculated and the average could be calculated over all the sources in the reach
>
> $$\lambda_x = \frac{\sum_{k=1}^{n} \min_k \{\lambda_{,i,j}\}}{n}$$ for *n* control variables, or
>
> Option 2: The individual attenuation acceptability could be aggregated conjunctively, in which case:
>
> $\lambda_x = \inf\{\min\{\lambda_{ij}\}\}$ for each stressor *i* and source *j*.

CONTROL VARIABLES.

The control variable need to express those entities that can be changed by the manager/ decision-maker in order to achieve the goal set in the objective function. There are two possible common denominators suggested by the objective function: the stressor levels and the degree of attenuation of the "raw" stressor levels. The advantage of the degree of attenuation is that it is unitless.

> The choice of control variable is the degree of attenuation of the "raw" stressor, designated by x. Each stressor *i* and source *j* combination is given a value x_{ij}.

CONSTRAINTS

The constraints describe the limits within which the optimisation must be performed. These might include physical constraints and process constraints. The might include the physical limitations on the value of the objective function, control variables or any other parameters involved in them. It may be required in the interest of being fair and equitable, that all similar stressors should be treated similarly.

The generic constraints chosen for this study are:

The attenuation levels by definition are defined such that $x_{ij} \in [0, 1]$ or $0 \leq x_{ij} \leq 1$.

The degree of satisfaction is defined such that $0 \leq \lambda \leq 1$.

It may be required that a maximum risk ρ is specified which may not be exceeded, therefore $\lambda_R \leq \rho$ and $0 \leq \rho \leq 1$

Optionally an equity constraint may be formulated such that for a stressor i from sources k and l the absolute difference between the attenuation of s from these source must always be less than an amount δ, i.e. $|| x_{ik} - x_{il} || \leq \delta$. δ is defined in Eq.[4.8]

Fuzzy constraints

- In order to produce such a general acceptability criterion, the user that may incorporate his own particular weighting of cost and technological implications of a treatment level x_{ij}. This requires at least an expression from each resource user of an acceptability pair $\{x_{ij}^{min}, x_{ij}^{max}\}$. Here, x_{ij}^{min} represents a treatment level that is completely acceptable, while x_{ij}^{max} represents a treatment level which, for whatever reason, is completely unacceptable.

- For this study it has been assumed that between these two levels (and possibly even including these levels) there exists a continuum of acceptability. Without loss of generality a stepped function could also have been used as long as the function is monotonic.

- Likewise, the regulator defines a fuzzy risk acceptability criterion by specifying (possibly resource dependent) *de minimis* and *de manifestis* risk levels, ρ^{min} and ρ^{max} respectively.

CALCULATION OF RISK/CONCERN VALUES

The ecological risk or concern, ρ, is calculated from the likelihood of the stressor occurrence and the cumulative likelihood of effect on exposure to a stressor. This requires either (1 OR 2) AND 3:

1. Measurement of the stressor values in-stream over a suitable spatial and temporal domain and estimating the likelihood of stressor occurrence from stressor observation data,
2. Modelling the stressor occurrence likelihood,

3. Estimating the stressor response likelihood from laboratory or field data.

Estimating stressor occurrence likelihood

Generally, the in-stream stressor value s_i will be a function of the unattenuated stressor value, s^0_{ij}, the treatment level, x_{ij}, the apparent stressor specific degradation constant, k_j, and the retention time τ_i between stressor entry point and the point of interest (see Appendix).

The ideal would be to estimate stressor occurrence likelihood from measured data. This is unlikely in the case of *ab initio* calculation of stressor attenuation. It is more likely that the second requirement can be met. Models of different levels of sophistication and environmental realism exist to calculate in-stream water quality parameters (e.g. CEAM, 1996). Predictive hydrological models also exist that estimate the in-stream flow from rainfall data (e.g. Pitman, 1973). Of the stressors selected for this study only the habitat degradation remains to be assessed *in situ*, but methods do exist to perform such an assessment (e.g. Kleynhans, 1996b).

For a probabilistic risk assessment, it is important that a stressor occurrence model be able to simulate the impact of temporal/spatial variability as well as model and/or parameter uncertainty. A common method to this is by Monte Carlo simulation. Possibilistic models would need to be able to deal with fuzzy inputs.

Two problems were encountered with the models that could be used for toxic substance models: 1) The software code for the models was not readily available, and 2) Few of the available models have the ability to accept or generate stochastic data. It was therefore difficult to integrate these models with rest of the coding used here. For the purpose of this study, a simple dilution model with constant first-order degradation kinetics was used to calculate the concentration of toxic substances, while it was assumed that the flow distribution was known *a priori*. A possibilistic model is described in 4.4.2. A stochastic analogue using Monte Carlo simulation was also attempted (coding appears in the Appendix of this chapter). This model was not pursued further for two reasons: the nature of the ecological impact favoured an epistemic approach to stressor occurrence that necessitated a possibilistic rather than a probabilistic methodology and the coding language used could not easily resolve the computer memory management problems encountered in the Monte Carlo simulation.

In most cases the stressor possibility distribution will be identical to the stressor variable distribution for example, in the case of toxic substances, the toxic stressor distribution will be identical to toxic substance concentration expressed as toxic units. However, in the case of flow, the flow itself is not the stressor, but flow insufficiency is more likely to be. In this case, the stressor possibility distribution derives from the extent to which the flow possibility distribution, $\mu_Q(q)$ can be said to be descriptive of the state of flow insufficiency, $\mu_I(q)$, and therefore: $\mu_S(q) = max\{min(\mu_Q(q), \mu_I(q)\}$. Here, the flow insufficiency is estimated from q^{min}, a level of flow below which organisms would likely succumb completely to the end-point effect and q^{max}, a level above which no end-point effect would be observable (see Appendix).

4.3.4 FORMULATIONS OF DSMS-OPTIMISATION PROBLEM

The conflicting needs of role players in a catchment was addressed by Tung (1992) in using multiple-objective WLA (involving the optimisation of conflicting needs to constraints) as an example of the application of multiple objective optimisation problems (MOOP's). Here the single objective concept of optimality is no longer valid. Unless a prior knowledge exists to weigh the conflicting objectives, the solution to the MOOP remains a locus of points representing a trade-off. The concept of optimality is replaced by the 'non-inferior solution' which is corresponds to a curve or surface until the decision-maker supplies the weighting. Chang *et al.*, (1997) applied fuzzy interval multiobjective optimisation to water pollution control in a river catchment showing that different types of uncertainty can be combined through a possibilistic approach. In general, these only consider water quality management in terms of discharge objectives.

In practice, the optimisation then involves finding the stressor and source specific treatment levels that maximises the acceptability parameter λ (or alternatively minimises the unacceptability $(1-\lambda)$)

CRISP FORMULATIONS

The optimisation problem may be formulated in several ways involving issues that may be of concern to the stakeholders, such as protection of the ecosystem, stressor reduction cost, and treatment equity among different stressor sources. From an ecosystem protection point of view the optimisation problem might be formulated as:

1. Minimise the cost of ecological concern (or risk) reduction by setting the stressor reduction level x_{ij} for the i^{th} stressor from the j^{th} source to a value that will satisfy an upper ecological risk limit for the system as well as possible technological or other ethical constraints.

2. Minimise the ecological concern (or risk) to the system by adjusting x_{ij} so as to meet cost, technological and ethical constraints.

3. Zimmermann's approach: maximise the degree of satisfaction of all stakeholder goals within given cost and risk constraints (Lai and Hwang, 1994).

$$
\begin{aligned}
&\text{maximise } \lambda \\
&\text{so that } \mu_R(x_{ij}) \geq \lambda && \text{(Regulatory goal)} \\
&\mu_G(x_{ij}) \geq \lambda && \text{(User goal)} \qquad [4.2] \\
&x_{ij} < \gamma_{ij} && \text{(Technological constraint)} \\
&|\,|\, x_{ik} - x_{il} \,|\,| \leq \delta && \text{(Equity constraint)}
\end{aligned}
$$

(Where γ_{ij} is the technological constraint for stressor *i* at source *j* and δ is a maximum allowable difference in attenuation level for stressor *i* between any two sources *k* and *l*).

In the first formulation it is assumed that it is feasible to estimate the financial cost as a function of x_{ij} quantitatively (Burn and McBean, 1985). Given that the unattenuated stressor magnitudes may in general be uncertain or variable, it would be necessary to set a compliance level α (say α = 0.95) and calculate the corresponding x_{ij}. The difference between the first two formulations of the problem is the aspect on which compromise has to be made. From a purely utilitarian point of view the second formulation is preferred while from a purely protective point of view the first formulation is preferred. However, both formulations require a functional relationship between constraints and control variables, but this is often lacking (Lai and Hwang, 1994).

FUZZY FORMULATION

A fuzzy set equivalent of this optimisation problem (Eq. [4.2]) could use the Bellman-Zadeh fuzzy decision (Z) which is defined as the intersection between fuzzy goals (G) and fuzzy constraints (C) (DuBois and Prade, 1994, Klir and Yuan, 1995), i.e. $Z = G \cap C$. This represents those goal and constraint values that satisfy both sets. The distinction between the goals and constraints is lost.

- The objective function supposes that each stakeholder will compromise on its constraint requirements and will be able to express its satisfaction with the consequence of a value of x_{ij} in terms of a satisfaction parameter λ.
- For resource protection, the protection agency may impose a risk level ρ_0, but will compromise that to the extent ρ'.
- Each stressor source may wish to reduce their expenditure for stressor reduction to a minimum. Each stressor source may set an ideal limit c_j but will compromise to the extent c'_j.

This translates the fuzzy programming formulation (Eq. [4.2]) to a crisp programming formulation (Eq. [4.3].

$$
\begin{aligned}
&\text{Maximise} \quad \lambda \\
&\text{So that } C_j(x_{ij}) \leq c_j + c'_j(1 - \lambda) \\
&R(x_{ij}) \leq \rho_0 + \rho'(1+\lambda) \qquad [4.3]
\end{aligned}
$$

$$x_{ij} < \gamma_{ij}$$

$$||\, x_{ik} - x_{il} \,|| \leq \delta$$

An interactive inexact fuzzy multiobjective programming (IFMOP), which is more extensive version of Eq. [4.3], was used (Wu *et al.*, 1997) in the water pollution control planning of a lake where the economic activities in the catchment had been specifically included. A problem that arose in this case related to separating objectives that had to be maximised from those that had to be minimised. In this case this difficulty does not arise since there is only one objective that needs to be maximised.

Application of the fuzzy formulation approach along with the constraints and terminology of 4.3.3 to Eq. [4.3] produces the model Eq. [4.4]:

Minimise (1-λ)

So that
$$\lambda = \begin{cases} 0 & if\, \lambda_R < \zeta \\ \min\{\lambda_R, \lambda_x, \lambda_{eq}\} & if\;\; \lambda_R \geq \zeta \end{cases} \qquad [4.4]$$

$$x \geq 0$$

and λ_R, λ_x and λ_{eq} as defined below in Eqs. [4.5], [4.6] and [4.7]. The parameter $\zeta \in [0, 1]$ is a minimum risk compliance level required by the regulator. The ecological risk with reference to the chosen level of organisation and end-point, ρ, is calculated from the possibility distribution of the stressor ($\mu_S(s_i)$) and the possibility distribution of the effect over the stressor range ($\mu_E(s_i)$).

The satisfaction terms in the optimisation model were calculated as follows:

$$\lambda_{ri,j} = \begin{cases} 1 & if\; \rho_{i,j} < \rho_{i,j}^{\min} \\ \dfrac{\rho_{i,j}^{\max} - \rho_{i,j}}{\rho_{i,j}^{\max} - \rho_{i,j}^{\min}} & if\; \rho_{i,j}^{\min} \leq \rho_{i,j} \leq \rho_{i,j}^{\max} \\ 0 & if\; \rho_{i,j} > \rho_{i,j}^{\max} \end{cases} \qquad [4.5]$$

$$\lambda_{ij} = \begin{cases} 1 & if\; x_{i,j} < x_{i,j}^{\min} \\ \dfrac{x_{ij}^{\max} - x_{ij}}{x_{ij}^{\max} - x_{ij}^{\min}} & if\; x_{ij}^{\min} \leq x_{ij} \leq x_{ij}^{\max} \\ 0 & if\; x_{ij} > x_{ij}^{\max} \end{cases} \qquad [4.6]$$

$$\lambda_x = \min\{\lambda_{ij}\}$$

$$\lambda_{eq} = \begin{cases} 1 & if \delta < \varepsilon_{\min} \\ \dfrac{\varepsilon_{\max} - \delta}{\varepsilon_{\max} - \varepsilon_{\min}} & if \varepsilon_{\min} \leq \delta \leq \varepsilon_{\max} \\ 0 & if \delta > \varepsilon_{\max} \end{cases} \qquad [4.7]$$

where $\rho = \max\{\min(\mu_S(s_i), \mu_E(s_i))\}$, $s_i = f(s_{ij}^0, x_{ij}, k_i, \tau_j)$ and

$$\delta = \max_i^m \left(\frac{\max_i^n\{x_{ij}\} - \min_i^n\{x_{ij}\}}{(\max_i^n\{x_{ij}\} + \min_i^n\{x_{ij}\})/2} \right) \qquad [4.8]$$

4.3.5 SOLVING THE OPTIMISATION PROBLEM

A large number of optimisation algorithms are available, of which two were selected as being conceptually simple as well as relatively easy to encode so that it could be effectively combined with a suitable objective function evaluation. The two that were eventually selected are the variable simplex and genetic optimisation algorithms.

THE VARIABLE SIMPLEX ALGORITHM

The Simplex algorithm (Nelder and Mead, 1965; Lowe, 1967; Betteridge, *et al.*, 1985; Gill *et al.*, 1991) is a heuristic search algorithm based on the projection of a simplex, which is a (*n*+1)-dimensional geometric figure for an *n*-dimensional search space. The objective function is evaluated at each of the *n*+1 vertices of the figure and a new figure is generated by projecting the worst vertex through the centre of gravity of the remaining *n* vertices. The Variable Simplex algorithm (Fig. 4.1) allows for contracting or expanding the projection in the Simplex algorithm to achieve a more rapid convergence to the optimum. Since this algorithm may be stuck at a local optimum, it is suggested that the search be restated at a different set of starting values. The algorithm as described by Shoup and Mistree (1987) was used.

GENETIC ALGORITHMS

Genetic algorithms (GA's) belong to the family of random search algorithms with a focussing heuristic (Bäck, 1996). GA's have as their basis the principles of Darwinian evolution. The mechanisms of GA's are similar to those in population genetics and are based on exchange of genetic material between individuals to produce new individuals whose suitability may differ from those of the parent individuals. The main operations are selection, exchange, mutation and reproduction. It is also possible to impose search heuristics to speed up the convergence. The version used here is of the elitist type where the best performing individuals are selected along with the offspring to compete in a tournament to find the best performing individuals.

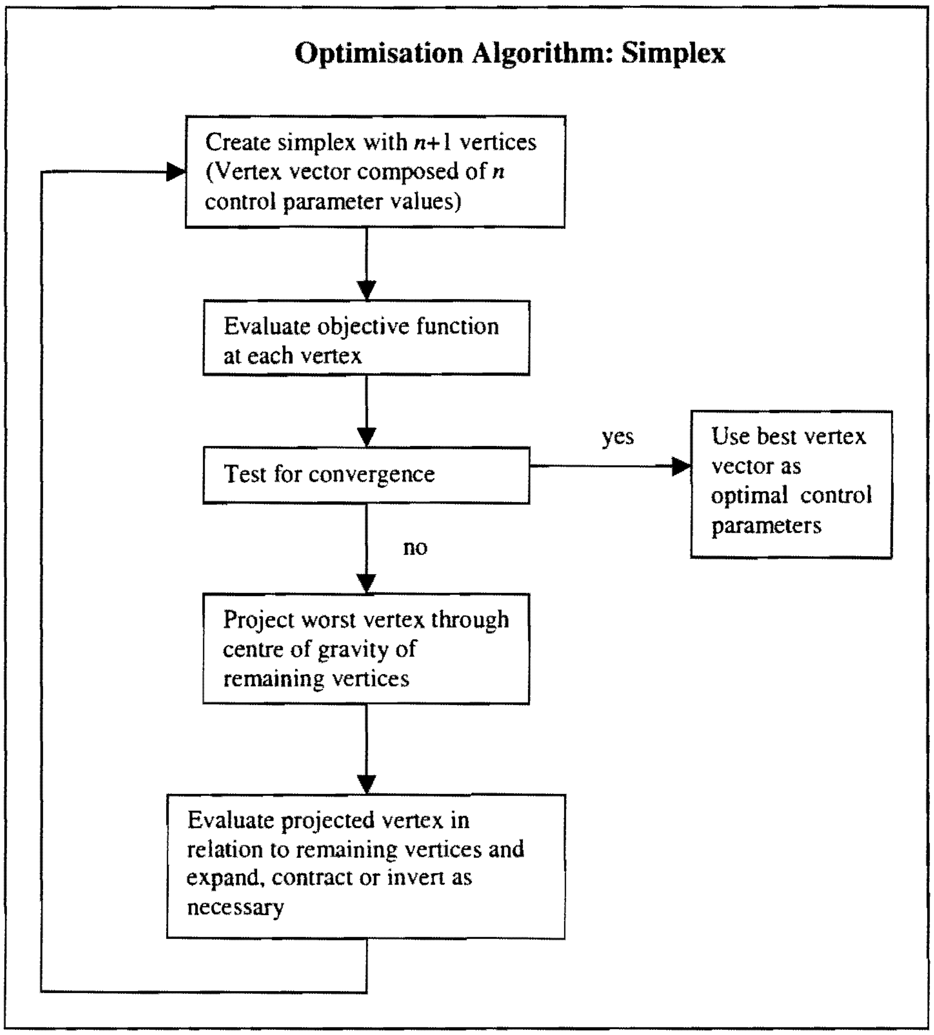

Figure 4.1. Diagram of the variable simplex algorithm.

4.4 HYPOTHETICAL CASE STUDY

The use of optimisation as a means to generate risk-based source criteria is investigated using a hypothetical test case. The parameters used in this case were not taken from any specific study, but represent considerations from a number of sources typical of situation in which such a method might be used.

The optimisation algorithms are first evaluated against a test (Colville response surface as described in Shoup and Mistree (1987)) where the optimum is known (Scenario 1). The genetic algorithm was then used to evaluate source specific criteria in three different scenarios resulting in different objective functions. In each of the last three scenarios two options for initialising the algorithm is evaluated. Some of the results are listed in the Appendix.

4.4.1 SELECTING STRESSORS AND SRR'S

The stressors chosen for the hypothetical case study are:

TOXIC SUBSTANCES

Although no specific general data on the occurrence of toxic substances in fresh water in South Africa were available, some problem related studies indicated that toxics do occur periodically in surface water. Chlorination is still a common practice on treated sewage effluent before discharge to surface water in South Africa (Williams, 1996). Toxicity assessments on chlorinated sewage from treatment plants in the KwaZulu-Natal province of South Africa indicated that it may an important contribution to surface water toxicity (Williams, 1996). The instream concentration of toxic substances will generally be a function of the input load to total load ratio, and will therefore be dependent on flow. It was further assumed that toxic concentration would be determined for point sources by a suite of whole effluent toxicity (WET) assessments. From the toxicity assessment data a concentration suitable to the end-point for the management goal will be selected e.g. a no-observed-effect concentration (NOEC) at a discrimination level $\alpha 1$. The level of the toxic stressor in the effluent, x, is expressed as toxicity units (TU's), which is calculated as: x (in TU's) = (the actual concentration of the effluent)/(NOEC) (Suter, 1993). The response curve for the risk assessment is simulated from the response curve from which the NOEC was calculated such the expected response y would be given by:

$$y = \frac{1}{1 + A_0 e^{-bx}} \qquad [4.9]$$

The constants A_0 and b are determined by solving [4.1] with the conditions that if $y = \alpha 1$ then TU = NOEC and if $y = \alpha 2$ then TU = $b2$ where $b2 = b$/NOEC and b is the concentration corresponding $\alpha 2$ in the original curve.

HABITAT DEGRADATION

Although no generic data were available for the South African status of instream habitat degradation as a stressor, some results (Sparks and Spink, 1998; Kleynhans, 1999b) seem to indicate that on a site-specific basis this may a major stressor to the aquatic ecosystem. Habitat degradation as a stressor must be distinguished from flow related habitat insufficiency, which was considered to be related to flow insufficiency (Milhous, 1998). As used here, habitat degradation refers to physical removal of aquatic habitat components, so that even when flow as represented by water depth or flow rate is sufficient, there is simply inadequate habitat to support aquatic life. No specific data on habitat stress assessment was found although the importance of habitat is recognised (Hardy, 1998; Lamouroux, *et al.*, 1998; Kleynhans, 1999a). The assessment of the response of aquatic organisms to physical habitat degradation has to be performed by a competent aquatic ecologist. The response curve may be estimated from a no-observable effect level of habitat degradation and an unacceptable level of habitat degradation corresponding to a threshold level below which no effect is expected and a level above which effects are certain to occur. The response may be simulated by a trapezoidal function or an s-shaped response from a function similar to Eq. [4.9].

FLOW INSUFFICIENCY

Water as the major habitat of aquatic organisms, needs to be maintained at a seasonally appropriate level for the aquatic ecosystem to remain functioning healthily (King and Louw1998; Moyle, *et al.*, 1998; Kleynhans, 1999b). In many cases the water depth is important as it provides access to specific habitat such as pools or riffles, which are important in the life histories of specific organisms. In some cases, the flow rate is important (Sparks and Spink, 1998). Flow insufficiency as a stressor does not include naturally occurring floods or droughts. Aquatic organisms in semi-arid countries may well have adapted to such events (Davies, *et al.*, 1994). Flow stress has, for the sake of illustration, been designated as (expected flow – actual flow)/(expected flow).

4.4.2 PROBLEM STATEMENT

Consider a river reach with three discharges and one abstraction. The magnitude of stream flow is representative of a small stream that already has significant toxicity present upstream of the reach being modelled. The discharges to this stream are typical of small sewage treatment works (about 1 megaliter per day). The toxicity, expressed as toxicity units, is based on chronic toxicity values and is not unlike those obtained for a small impacted stream in an industrialised area in South Africa (Jooste and Thirion, 1999). The habitat stress is assumed to derive mostly from streambed modification through farming and construction activities. Although streams of this magnitude are not significant as major water suppliers, they are typical of those that may be the refugiae and possible sources of recolonisation for larger streams and rivers and may be worthy of being protected for this reason.

The stream is modelled as a system with four nodes (see Figure 4.1) with inputs and outputs. The first two nodes receive discharges, the third node yields abstraction and the fourth node receives discharge. The habitat stress is associated with the node upstream of the stressed habitat.

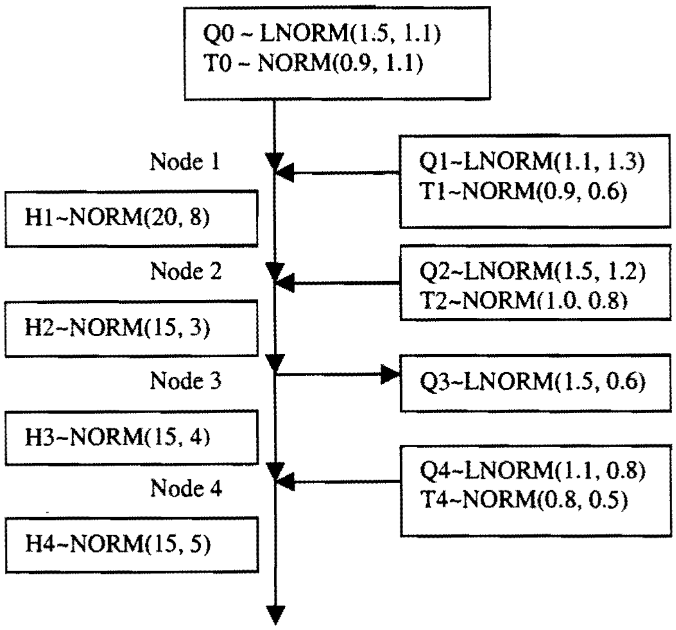

Figure 4.1 A diagram illustrating the set-up of the hypothetical test case. The input values for the stochastic problem formulation are shown. For a median x and standard deviation y, LOGNORM(x,y) indicates the lognormal distribution and NORM(x,y) indicates a normal distribution.

The control variables are:

1) the attenuation of the volume of water abstraction (xQ3),
2) the attenuation of toxic substances at the discharge nodes (xT1, xT2 and xT4) and
3) the habitat stress attenuation at each node (xH1, xH2, xH3, xH4).

The discharge flows, the discharge toxic concentrations, the habitat stressor levels as well as the upstream flow and toxicity levels are considered stochastic variables. It is assumed that the toxicity in the river is subject to degradation following a simple exponential decay function. The toxic levels at each node are calculated by mass balance (Eq. [4.10]).

$$sd_{i,j} = \frac{su_{i,j} \cdot qu_j + (-1)^{z_j} \cdot q_j \cdot (1 - f_{1,j})^{z_j} \cdot \{s_{i,j-1} \cdot z_j + s_{i,j} \cdot (1 - f_{i,j}) \cdot (z_j - 1)^{z_j}\}}{qd_j}$$
$$qu_j = qd_{j-1}$$
$$qd_j = qu_j + (-1)^{z_j} \cdot q_j \cdot (1 - f_{1,j})^{z_j}$$
$$su_{i,j} = sd_{i,j-1} \cdot \exp(-k_{i,j-1} \cdot \tau_{j-1}) \qquad [4.10]$$

where $i \in$ {dilution dependent stressors}, $j \in$ {sources}, f_{ij} is the attenuation factor and $z_j = 1$ for an abstraction and 0 otherwise. In the hypothetical case $i \in$ {T} and $j \in$ {1, 2, 3, 4} and $z_3 = 1$. For control variables $f_{ij} \in [0, 1)$ (i.e. $f_{ij} = x_{ij}$), while for non-control variables $f_{ij} = 0$.

The in-stream habitat degradation values remain unaltered over time but can be attenuated.

Table 4.1. Numerical input values for the model described in the text (Scenario 2).

Parameter	Upstream	Point 1	Point 2	Point 3	Point 4
Flow median q^0	4	1.1	2.1	2.5	1.8
Flow std dev.	1.1	1.15	1.28	1.56	1.11
Tox units median t^0	0.3	0.8	1.1	0	0.9
Tox units std dev	0.1	0.21	0.34	0	0.26
Habitat degr. Min	0	10	15	10	20
Habitat degr. Med	10	20	30	30	30
Habitat degr. Max	20	30	40	50	50
Q^{min}	-	1.5	1.5	2	2
Q^{max}	-	2.1	2.1	2.5	2.5
Flow stress effect min	-	2.2	2.5	2.5	2.6
Flow stress effect max	-	3.5	3.5	3.5	4
Tox stress effect min	-	0.2	0.2	0.2	0.2
Tox stress effect max	-	1.1	1.1	1.1	1.2
Habitat stress effect min	-	30	30	30	30
Habitat stress effect max	-	75	75	65	75
Tox degradation constant k (day^{-1})	-	0.2	0.2	0.2	0.2
Retention time τ (days)	-	2	3	2.5	4
Treatment acceptability					
x_q^{min}	-	-	-	0	-
x_q^{max}	-	-	-	0.6	-
x_t^{min}	-	0.2	0.2	-	0.3
x_t^{max}	-	0.7	0.8	-	0.75
x_h^{min}	-	0	0	0	0
x_h^{max}	-	0.15	0.1	0.1	0.2
Regulator risk acceptability					
ρ^{min}	-	0.05	0.05	0.05	0.05
ρ^{max}	-	0.15	0.15	0.15	0.15

4.4.3 METHODOLOGY

GENETIC ALGORITHM

In order to create the genetic material, the initial values for the control parameters in the optimisation problem were encoded as 16-bit binary numbers. All values were multiplied by 1000 and truncated to integers. The gene characterising an individual was created by the concatenation of the 16-bit binary numbers.

The genetic algorithm is outlined in Fig. 2. In the genetic algorithm, the vector of control parameters was considered as a part of a "chromosome" characterising an "individual" solution to the optimisation problem. The control parameter values were multiplied by 1000 to give a value in the interval [0, 1000]. This was done in order to facilitate the conversion of control variables to binary format.

From these two parents an initial population of 16 individuals (including the parents) were generated each with its own chromosomal values, by methods as described in the Appendix. These were then converted to binary numbers and encoded into a 16-bit string for each of the control parameters. The genetic algorithm used in this study was of the "elitist" type where the four best parents were preserved as part of the next generation. The parents were selected randomly with an exponential probability distribution (location parameter = 1).

The crossover was selected so that each 16-bit byte had an equal chance of being selected from either parent. Mutations, where the 0's and 1's were inverted on transcription of the parent bit to the child bit, were performed with a probability of 0.1.

The performance of the each individual in the population was determined by decoding the chromosome into control parameters and recalculating λ. The population was then rearranged from best to worst, based on the λ values.

After every epoch of 40 generations the control parameters were re-initialised from a suitable distribution and this process was repeated for 10 epochs. This cycle was repeated 10 times.

The performance of the best individual in the population was recorded, as were the values of the control parameters corresponding to the best performing individual in the population. In order to speed up the process both the range of the search domain and a heuristic adaptation the direction of search for each control variable was performed after every 5 generations (Ndiritu and Daniell, 1999). After refocusing and adaptation the population was reinitialised.

Methods used for the assignment of control variable values in the genetic algorithm:

(a) For initialisation, two parent individuals are generated by random assignment of control variable values from the interval [0, 1] by different distributions. The individuals are selected on the basis of producing a value $(1-\lambda) < 1$. The control variable values for the initial population are generated from the parent values by the random addition of $\pm$(0.3* the parent value) to the parent value.
(b) For the re-initialisation of control variable values after each epoch or after refocusing, the tournament population was generated by assigning the values from the variable specific interval $[x_i^{min}, x_i^{max}]$ by exponential distribution with location parameter μ where $\mu = 2\ln(0.5)/(x_i^{max} - x_i^{min})$.

The two options in assigning the control variable values in the initialising and re-initialising steps are:

- Option 1: initialise from a uniform distribution and re-initialise from an exponential distribution and
- Option 2: initialise and re-initialise from an exponential distribution.

TESTING OF ALGORITHMS

The performance of both algorithms were tested by obtaining the minimum of the four parameter Colville response surface described by Shoup and Mistree (1987).

The fundamentals of the methods for the Variable Simplex and GA used are described in Shoup and Mistree (1992) and Ndiritu and Daniell (1999) respectively. The coding of the methods was tested by using the Colville response surface and establishing whether the optimum point could be reached.

Table 4.2 Parameters for the evaluation of coding for the simplex and genetic optimisation algorithms.

Parameter	**Value**
Simplex: Expansion coefficient α	1.0
Contraction coefficient β	0.5
Contraction coefficient γ	0.5
Genetic algorithm	
Number of cycles (s)	10
Number epochs per cycle (e)	10
Number of generations per epoch (g)	40
Number generations for focussing (g1)	5
Number of generation for heuristic shift (g2)	5
Probability of mutation (m)	0.1

The hypothetical test case was then coded in Microsoft® QBASIC and run on a 333 MHz PentiumII processor with parameters as set out in Table A4.2 in the Appendix to Chapter 4. For the genetic algorithm, the basic algorithm and attempted improvements as well as the respective coding appear in the Appendix.

Both simplex and genetic algorithms found the theoretical extremum within about 50 iterations. However, application of the simplex algorithm failed to converge in the hypothetical case above.

CALCULATING STRESSOR VALUES

The procedure followed in the calculation of point source stressor attenuation values is outlined in Fig. 4.2. The characteristics of the three sources of discharge and one abstraction are shown in Table 4.1 (Scenario 2). The calculations were repeated with two other scenario's where the acceptability range for Source 1 was changed to $x \in (0, 0.3]$ (Scenario 3) and another where the risk acceptability was changed to $\rho \in [0.01, 0.05]$ (Scenario 4).

Generating possibility distributions

Instead of treating the inputs to the mass balance equation (used to calculate the toxicity levels from stochastic inputs) as a stochastic quantity, it was interpreted as a deterministic variable that is subject to epistemic uncertainty. For the purpose of this calculation the probability distributions were treated as possibility distributions by normalising to the maximum of the probability distribution (i.e. the possibility that $X = x$, $\Pi(X{=}x) = P(X{=}x)/P(X{=}\text{ mode } x)$).

The calculation of the fuzzy toxicity level was then performed by considering nested sets of intervals based on α-cuts of the stressor possibility distributions (Kaufman and Gupta, 1985; Klir and Folger, 1988), using interval arithmetic (Alefeld and Herzberger, 1974). The possibility range of each variable was discretised into 20 values (including 0 and 1). The upper and lower bound toxicity levels were calculated at each α-level, which corresponds to an upper and lower risk level. The risk satisfaction level ,λ_R, was calculated from the maximum risk and the risk acceptability values ρ_{min} and ρ_{max}. In order to counter the possible degeneracy induced by the fuzzy objectives in Eqs.[4.5] and [4.6], values ρ^{min} and ρ^{max} and x_{ij}^{min} and x_{ij}^{max} were used as the abscissa values corresponding to the ordinate values of 0.05 and 0.95 respectively in Eq. [4.9], while q^{min} and q^{max} were used as the abscissa values corresponding to the ordinate values 0.95 and 0.05 respectively in Eq. [4.11].

$$y = \frac{A_0 e^{-bx}}{1 + A_0 e^{-bx}} \qquad [4.11]$$

The control parameters were selected as those attenuation values that were actually controllable. The abstraction concentration and the effluent flow attenuation were not considered to be practically controllable. This resulted in eight control parameters being used, i.e. $x_k \in [0, 1]$, $x_k \in \{f_{ij}\}$, $i \in \{Q, T, H\}$ and $j \in \{1, 2, 3, 4\}$ for the test case.

ESTIMATING THE INFLUENCE FUZZIFICATION PARAMETERS

To estimate the effect a change in acceptability parameters will have the toxic attenuation acceptability parameter for source 1 and risk acceptability parameters were adjusted as shown in Table 4.3.

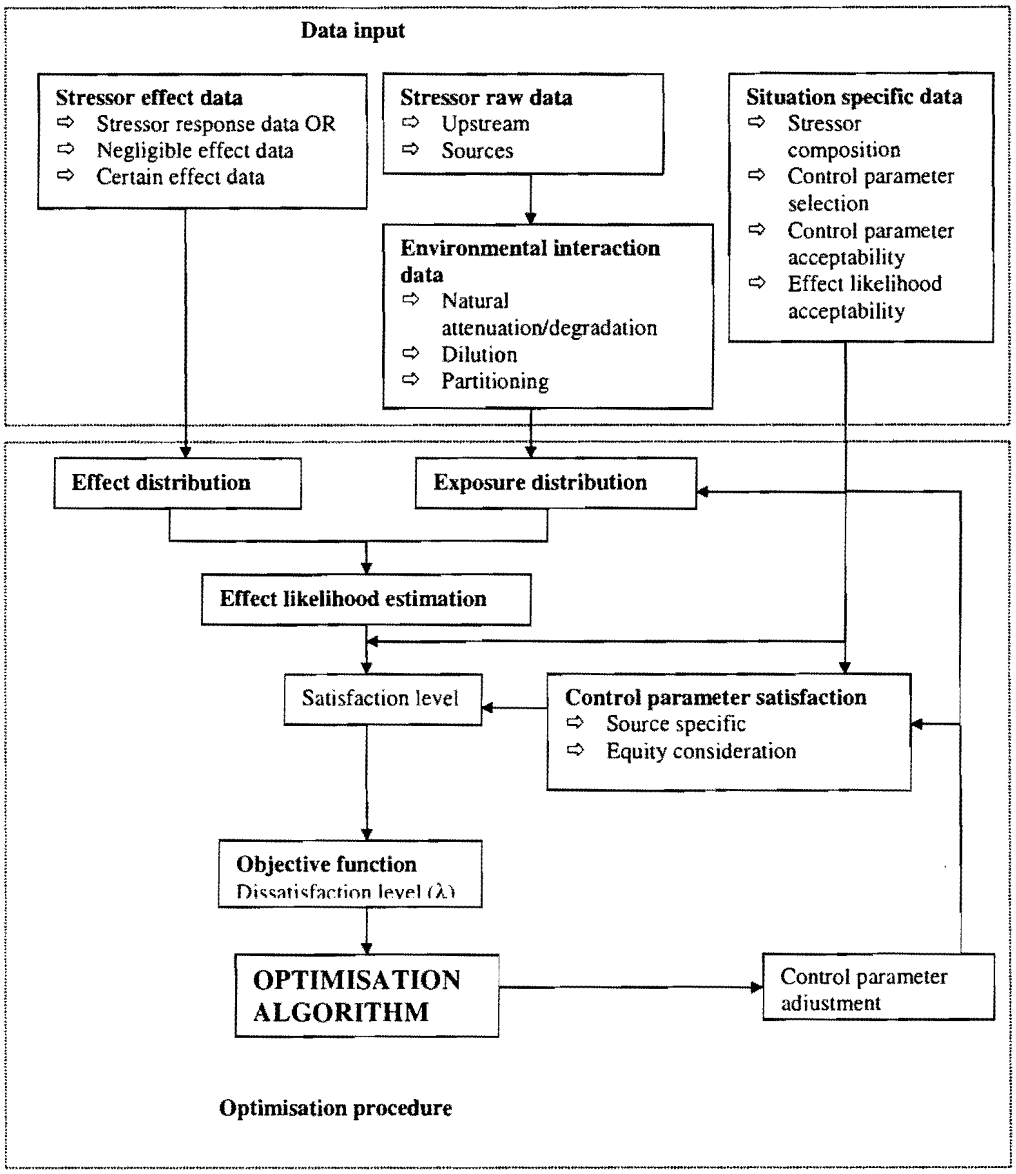

Figure 4.2. An outline of the methodology used to calculate the stressor attenuation levels.

Table 4.3. Acceptability parameter values for Scenarios 2, 3 and 4

Scenario	$x_{t,1}^{\min}, x_{t,1}^{\max}$	$\rho^{\min}, \rho^{\max}$
2	0.2, 0.7	0.05, 0.15
3	0.01, 0.3	0.05, 0.15
4	0.2, 0.7	0.01, 0.05

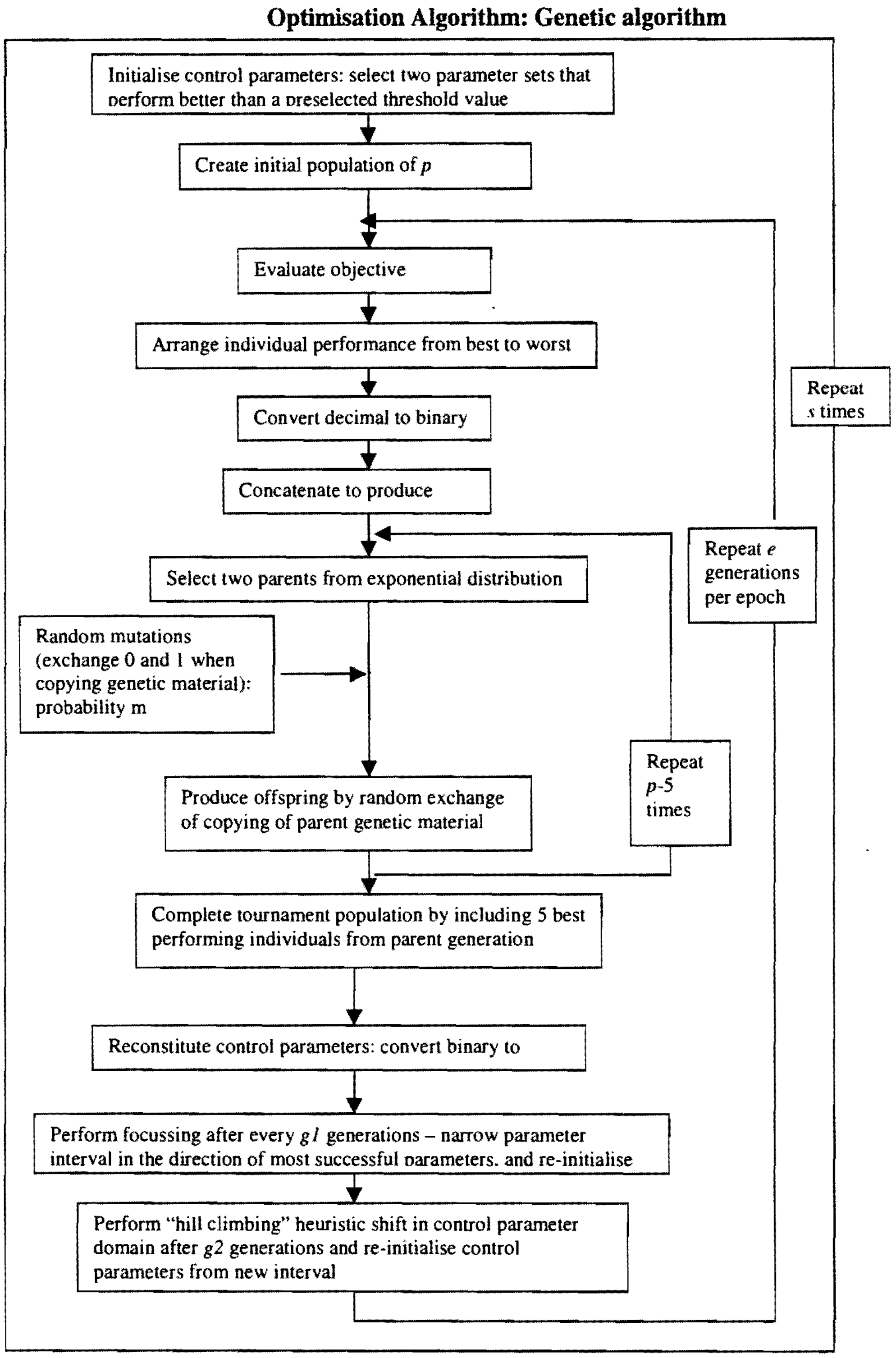

Figure 4.3. An outline of the genetic optimisation algorithm used in the estimate the attenuation levels for multiple stressors.

4.4.4 RESULTS

COMPARISON OF ALGORITHMS

The results for the comparison between the Variable Simplex and GA optimisation appears in Table 4.4.

Table 4.4 Results of the coding tests on the Colville response surface.

Algorithm	Result
Variable Simplex	Convergence dependent on choice of initial values. With favourable choice of initialising values converges in 40 to 50 iterations to within 10 in 1000 000, i.e. about 200 to 400 evaluations of objective function. One hundred repetitions of the process with random initial values did not produce one case of convergence.
Genetic Basic	Convergence independent of initial values if total number of generations > 100 and initial population ≥ 4*number dimensions, i.e. > 2 000 evaluations of objective function. Ten repetitions of the process produced six cases of convergence. (Parameter values found by trial and error.)

The result for the Variable Simplex algorithm is different from that obtained by Shoup and Mistree (1987) who obtained convergence for the Colville response surface irrespective of the initialising values of the control parameters. The reason for this difference is not immediately apparent. It was assumed that some coding error must have caused this difference, but meticulous checking of the coding did not reveal an obvious error. Although the variable simplex algorithm outperformed the genetic algorithm on the Colville response surface in terms of the number of iterations needed in order to obtain convergence, the dependence of the convergence on the initial values was considered enough reason not to investigate the use of the variable simplex in the catchment optimisation problem. Early attempts at using the variable simplex algorithm on the catchment problem showed that there was no convergence in control parameter values after 400 iterations. Consequently, despite its computational expense, it was considered necessary to use the genetic algorithm approach for the catchment optimisation problem.

The in-stream toxicity stressor values generated by the α-cut method and the corresponding effect expectation values are shown in Fig. 4.4. The first two trials involved a comparison of the choice of initialisation option with the use of the average minimum aggregation for λ_{x}.

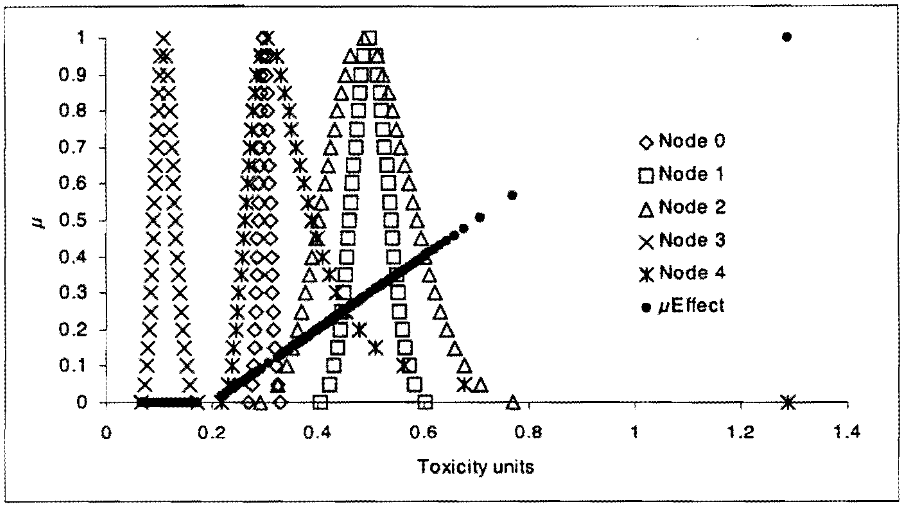

Figure 4.4. The possibility distribution of toxics as calculated at each node before attenuation. By way of comparison the toxic effect membership values used in the calculation are also plotted.

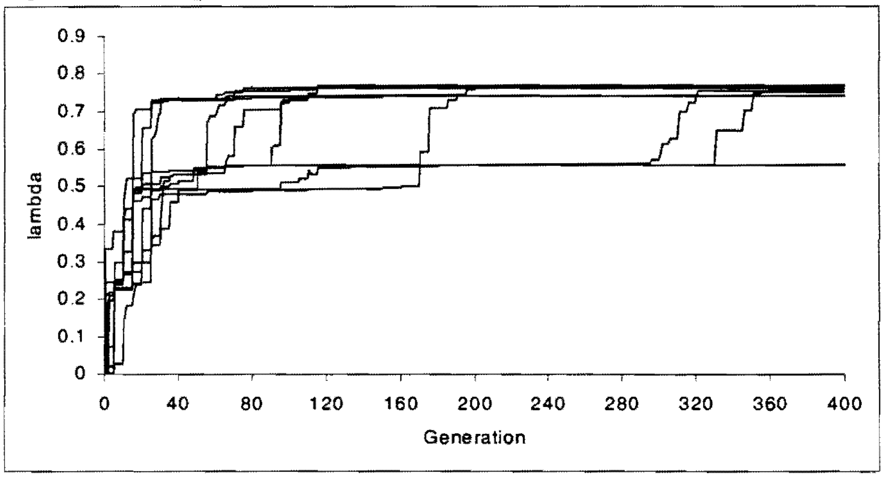

Figure 4.5. The best λ as function of the number of generations per cycle with Option 1 using the average minimum aggregation for λ_∞.

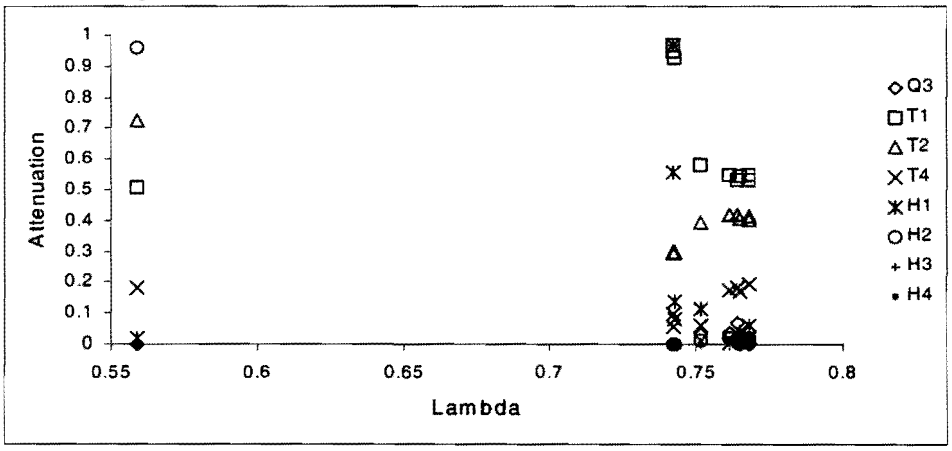

Figure 4.6. The attenuation values corresponding to the best λ in each cycle with Option 1 using the average minimum aggregation for λ_∞.

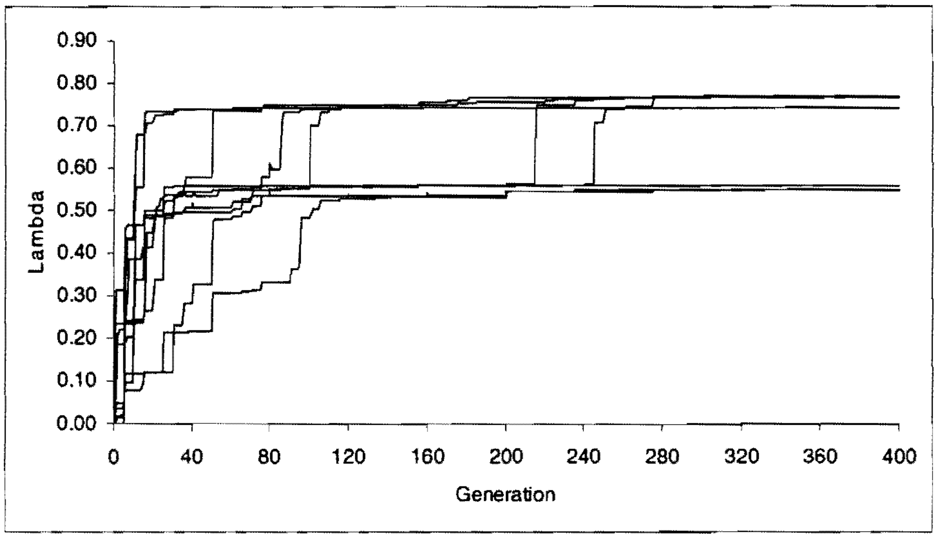

Figure 4.7. The best λ as function of the number of generations per cycle with Option 2 using the average minimum aggregation for λ_v.

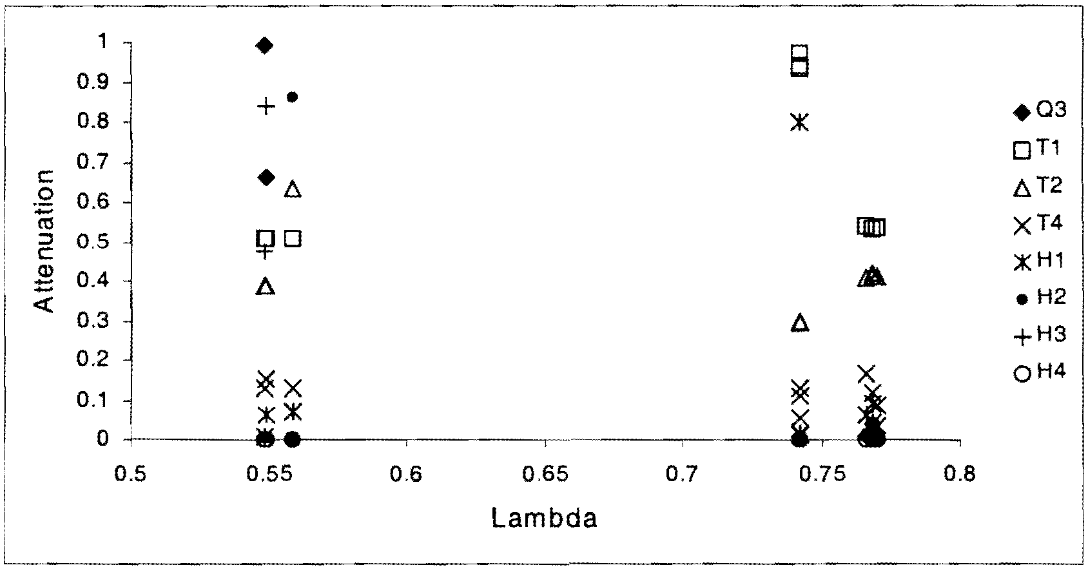

Figure 4.8. The attenuation values corresponding to the best λ in each cycle with Option 2 using the average minimum aggregation for λ_v.

The convergence rates of λ are compared in Figs. 4.5 and 4.7. Both Figures show that there are probably two minima: one with λ = 0.72 and the other with λ = 0.54. Option 2 (both population initialisations from exponential distributions) shows a marginally better convergence rate then Option 1. Comparison of Figs. 4.6 and 4.8 shows the optimal attenuation vectors for the two options compare well.

Toxicity attenuation requires the most attention, as can be expected from the possibility distributions, with source 1 requiring the highest attenuation. This corresponds well, with the intuitive notion that the relatively high toxicity and habitat degradation values at node 1 will result in an increased overall risk just downstream of node 1. The flow and habitat stressors need little attenuation ($x_{i,j}$ < 10%).

The attenuation values in Figs. 4.6 and 4.8 show discrimination among identical stressors (e.g. toxics) as well as raising the issue of neglect of specific source satisfaction. Here, average minimum aggregation may well balance a zero satisfaction at one source with a higher satisfaction at another source. This might argue for applying minimum satisfaction aggregation of individual stressor satisfaction.

When both minimum satisfaction aggregation and equity constraints are applied to the Option 2 algorithm, the results in Figs. 4.9 and 4.10 are obtained. This shows that the convergence rate of the algorithm has slowed down significantly so that in 400 generation the best satisfaction,λ, was only about 0.15. The stressor attenuation appears satisfactory from an equity point of view but it was attained at the cost of higher flow-stressor attenuation.

The lower overall λ might suggest that this application places an unfair burden on stressor sources. The question is if the imposition of risk constraints is the cause of the lower λ. Comparison of Figs. 4.11 and 4.12 with Figs. 4.6 and 4.9 would suggest that λ be dominated by λ_x. Other data (shown in Appendix 4) indicated that the risk satisfaction level, λ_r, is highly variable but in the runs corresponding to Figs 4.11 and 4.12, $\lambda_r \in [0.78, 0.99]$ and $\lambda_r \in [0.16, 0.99]$ respectively. This would seem to indicate that while the risk constraints might steer the control variable selection in the direction of lowest λ_x.

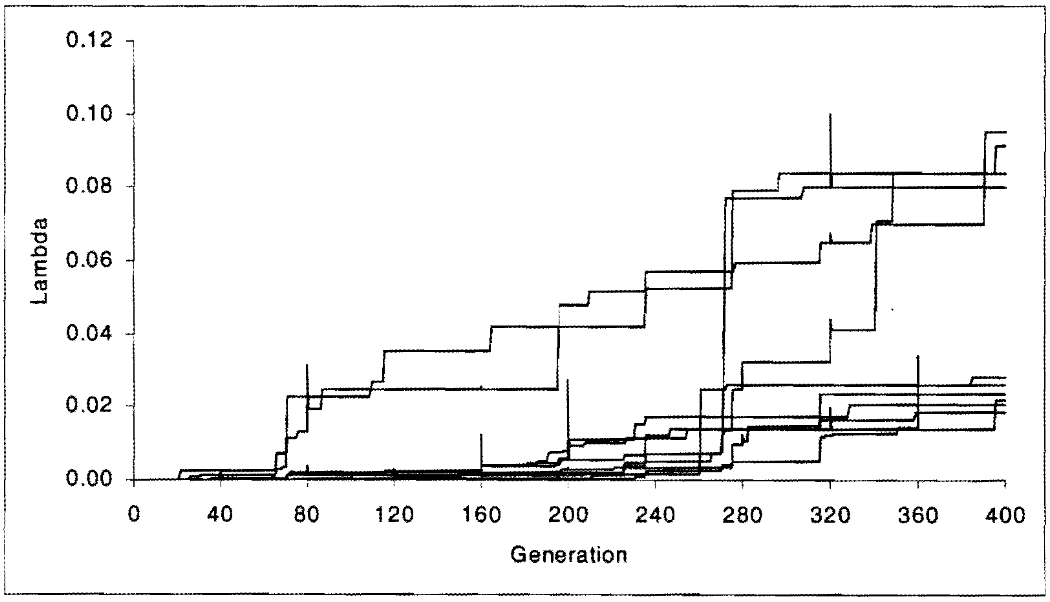

Figure 4.9. The best λ as a function of number of generations in a cycle with Option 2 and including disjunctive aggregation for λ_x and stressor specific equity constraints.

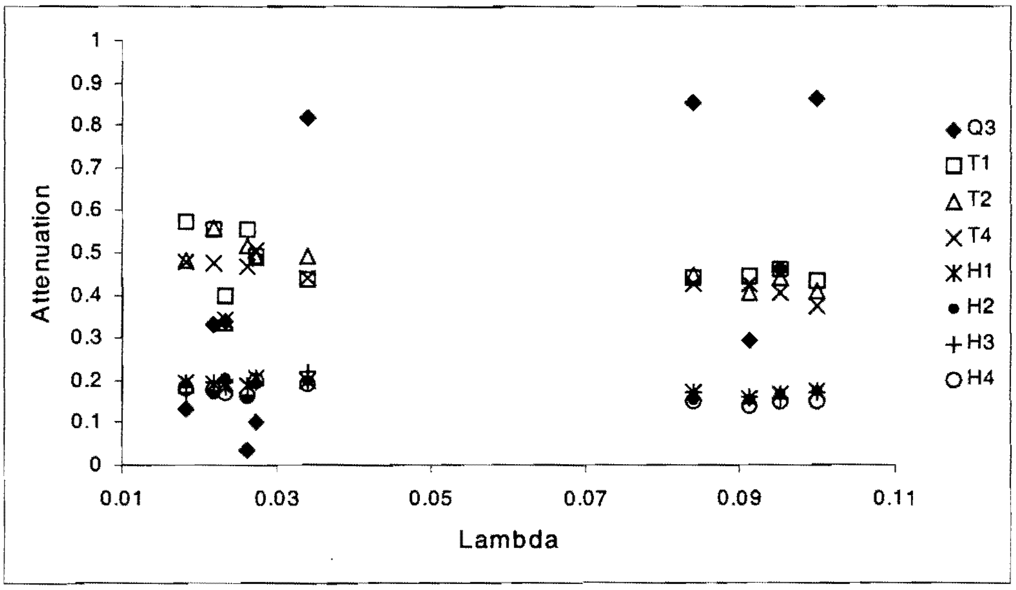

Figure 4.10. The attenuation values corresponding to the best λ per cycle with Option 2 and including disjunctive aggregation for λ_x and stressor specific equity constraints.

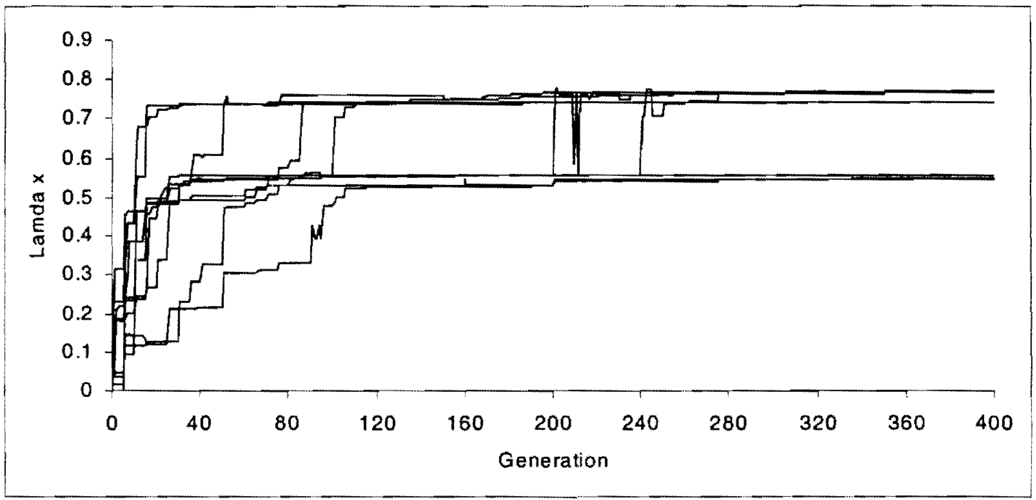

Figure 4.11. Attenuation satisfaction λ_x as a function of the number of generations with Option 2 and average minimum aggregation for λx (no equity constraints). Comparison with Figure 4.7 shows that λ is dominated by λ_x.

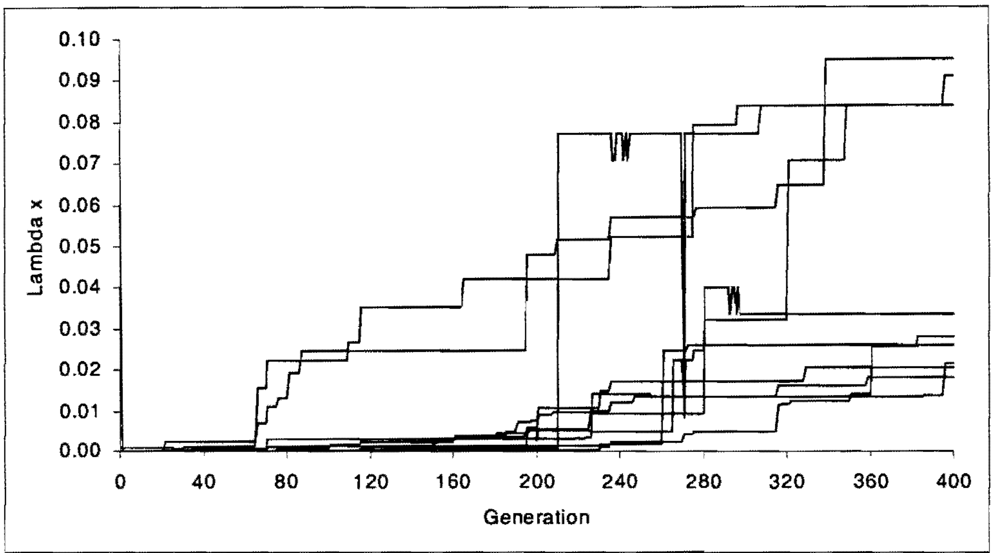

Figure 4.12. Attenuation satisfaction λ_x as a function of the number of generations with Option 2 with disjunctive aggregation for λx and stressor attenuation equity constraints. Comparison with Figure 4.9 shows that λ is dominated by λ_x.

The stressor attenuation values predicted by this algorithm are listed in Table 4.5.

Table 4.5. Stressor attenuation values for various algorithm options.

	Option 1	**Option 2**			
Equity constraint	No	No	No	Yes	Yes
λ_x aggregation	Av. Min.	Av. Min.	Conjunctive	Av. Min.	Conjunctive
x_{T1}	0.039	0.004	0.141 – 0.914*	0.289	0.461
x_{T2}	0.549	0.563	0.689 – 0.957*	0.497	0.461
x_{T4}	0.404	0.410	0.542 – 0.993*	0.522	0.440
x_{Q3}	0.062	0.086	0.023 – 0.167*	0.515	0.405
x_{H1}	0.060	0.037	0.047 – 0.833*	0.071	0.169
x_{H2}	0.004	0.160	0.067 – 0.915*	0.068	0.168
x_{H3}	0	0	0.043 – 0.964*	0.070	0.159
x_{H4}	0	0	0	0.074	0.149

* Variable attenuation values with a degenerate λ = 0.99

The computation time for this optimisation could be significant. An optimisation code written in Microsoft® QBASIC (in which the development was done) running on a 333 MHz Pentium II processor took between about 3 hours to complete the optimisation. While it is recognised that substantial computation time saving can be brought about by more efficient coding, computation time is likely to remain significant. However, in comparison to the time required to perform stochastic WLA's, this time expenditure is probably not excessive.

4.4.5 DISCUSSION

Determining the source specific stressor-attenuation values by the optimisation of ecological concern to process-related acceptability appears a viable method to arrive at site or situation specific management criteria.

In the example used above, it has tacitly been assumed that the methodology exists by which the stressor-specific response curves can be generated. In all cases, this would involve a significant amount of effort. In most cases such methodology is not readily available or is still subject to development.

In the case of toxics, recourse will likely have to be taken to ecotoxicological data. However, the common laboratory scale LC50 or EC50 data on its own, is hardly likely to suffice. The selection of the correct metric to represent the ecosystem-level effects is a subject for expert deliberation based on system specific knowledge.

In the case of flow related stress, it seems feasible that some of the developments currently under way on the estimation of in-stream flow requirements (e.g. King and Louw, 1998) could eventually be used to parameterise the flow-stress response relationship.

Habitat stress response is likely to be an expert-input driven assessment and the level of input very similar to that of a risk assessment. In fact, the input required for each stressor is virtually the same as for the effect assessment phase of an ecological risk assessment of each stressor.

While the data and information requirements of this approach are high, the potential exists for each water user (where "use" is defined not only in terms of abstraction but also as discharge) as well as the regulator to effect compromises. At the same time the water users are required to consider their requirements carefully. Although simple trapezoidal acceptability functions were used in this example, these functions could be quite complex, without detriment to the overall process.

The risk objective values clearly have a significant impact on the attenuation values estimated by this procedure (Appendix 4, Figs A4.4.5 and A4.4.6). It can make a very dramatic difference in the attenuation of toxics at source 1, with resultant cost and other implications. Careful attentions need to be given to the derivation of these values so that they correlate to field observations such as biomonitoring results.

Given the complexity of the process in deriving the information necessary to perform this optimisation, it is unlikely that this approach to stressor attenuation calculation will be used at a primary level. A typical application scenario would require that a hazard-based screening tier would precede the use of this model. As the rate of return of environmental benefits slows down when increasingly strict effluent standards are applied, a critical appreciation of effect-based models (such as the ecological concern model used here), will become increasingly important (Somlyódy, 1997). Affordability in river basin management can be addressed by the combined use of effluent criteria (as a minimum requirement) and ecological risk or concern objectives as means to refine and adapt such criteria.

Structure

The document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

Part 3: (This Part) Presents some of the papers that have been published in peer reviewed literature and that are included for quick reference.

Part 3:

Technical Papers

Rationale for an ecological risk approach for South African water resource management

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Abstract

The principle of ecosystem protection in the South African Water Act requires that water resource management tools for a multiple stressor environment be tailored to the characteristics of the aquatic ecosystem. The requirements of the Act, the characteristics of aquatic ecosystems as well as co-occurrence of diverse stressors are considered. Although single substance criteria have a useful role, they are not sufficient for resource management within the context of the ecological reserve. It is proposed that an effect-likelihood approach has the potential to address the variability and uncertainty in management of a surface water body subject to multiple stressors. An in-stream receiving water risk objective approach might be considered.

Glossary

ERA	Ecological risk assessment
Hazardous	Having the potential to cause an (undesired) effect.
IFR	In-stream flow requirement
SAWQG	South African Water Quality Guidelines
Stressor	An anthropogenic substance, form of energy or circumstance that may cause a loss of sustainable ecosystem function.

Introduction

The South African national water policy considers the aquatic ecosystem to be an integral part of the resource base from which water is derived for human and environmental use, but "only that water required to meet basic human needs and maintain environmental sustainability will be guaranteed as a right. This will be known as the Reserve" (DWAF, 1997). This concept was also embodied in the National Water Act (NWA, 1998). The environmental or ecological aspect of the reserve has been identified in such a way that it must ensure water quantity and water quality which are appropriate to meet these needs. The term resource quality "is used to include the health of all parts of the water resource, which together make up an 'ecosystem', including plant and animal communities and their habitats" (DWAF, 1997).

This paper presents a rationale for the use of ecological risk in water resource management in South Africa within the context of the NWA.

Background

Two distinct philosophical approaches that can be applied to water resource quality management are summarised in Table 1.

While the approaches in Table 1 are presented as extremes in philosophy, there is a growing appreciation for the need for, and a movement toward, a holistic, integrative approach in environmental management generally and water resource management in particular (e.g. Foran and Fink, 1993; EEC, 1994; Schneiders, et al., 1996; USEPA, 1997). Such a holistic approach to water resource management strongly features sustainability linked to some ecological entity (or objective) (e.g. CUWVO, 1988; Wils et al., 1994; Schneiders et al., 1995; USEPA, 1997). The ecological objectives then become either directly or indirectly the basis of, for example, water quality criteria. Ecological risk methodology can be applied to both extremes and an integrated approach and does not stand in contrast to any of these approaches.

A proposal for the application of ecological risk to the ecological reserve is shown in Fig. 1. The rationale of using ecological risk concepts in water resource management is based on three observations:

- the implications of aspects of the NWA as indicated above,
- the "diverse stressor problem" and
- the inherent characteristics of aquatic ecosystems.

Implications of the NWA

It is implicitly recognised that use of the resource is not only allowed, but is also necessary for the well-being of the country and that this use needs to be managed in a way that will ensure sustainability. In this context it is noted that:

- The terms "use" refers not only to consumption and recreational use, but also to discharge of anything that may affect, *inter alia*, the sustainability of use.
- The NWA makes provision for protective measures for the water resource which includes classification of the resource and setting resource quality objectives that will give effect to the reserve set for that class.
- The ecological component of the reserve refers to a quantity and quality of water that will ensure ecologically sustainable development of the resource.
- Resource quality includes the quantity, pattern, timing, water level and assurance of in-stream flow, the physical, chemical and biological characteristics of the water, the character and condition of the in-stream and riparian habitat as well as the characteristics, condition and distribution of the aquatic biota.

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TABLE 1
A comparison of a technology-based and an ecological effect-based approach to resource management

Aspect	Technology-based approach	Ecological effect-based approach
Point of departure	Technology determines the best attainable stressor levels.	Ecological effect determines the most suitable stressor levels
Characteristic expressions	Best available technology (BAT); Best available technology not entailing excessive cost (BATNEEC); Best management practice (BMP); Best practical technology (BPT), etc.	"Fishable and swimable rivers"; "protecting most species most of the time", "maintaining sustainable ecological function", etc.
Main advantage	Proven technological feasibility.	Directly related to environmental goals
Main disadvantage	Environmental impact largely retrospective.	Required stressor levels not necessarily feasible or viable.

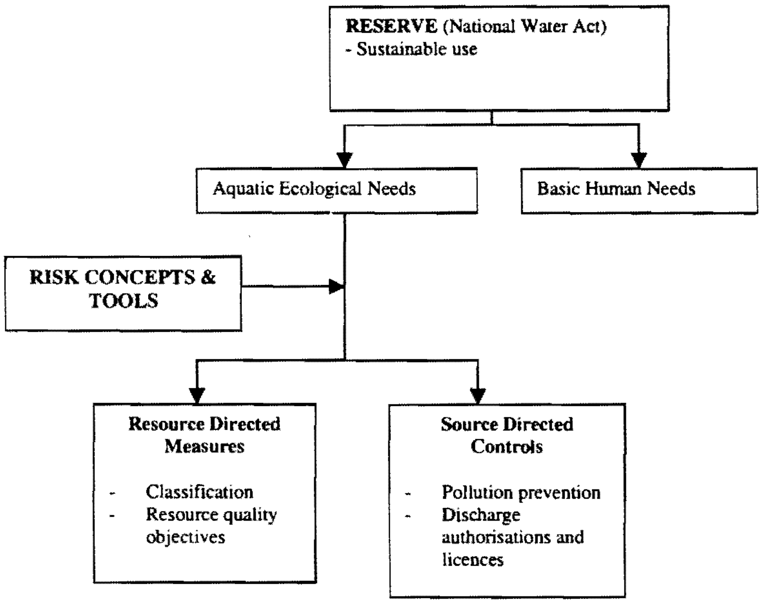

Figure 1
The potential inputs of ecological risk methodology to aspects of water resource quality management.

It is recognised that some activities that may cause stress to the aquatic ecosystem will have to be allowed, but that these have to be controlled in a manner that allows ecological sustainability.

Furthermore, the NWA differentiates between classes of resources, which correspond to a differentiation in some aspect of sustainability. Risk to the resource base was proposed as the basis of differentiation (DWAF, 1997). Here, irreversible damage to the resource base approximates a loss of sustainability.

Consequently, although the term "risk" does not appear explicitly in the NWA as the basis for classification, implicitly it is recognised that different classes of a resource will be subject to different degrees of risk of unsustainability and, by implication, different activities will result in different levels of risk.

The diverse stressor problem

Water use may entail a change in resource characteristics such as chemical composition, physical characteristics, flow and water depth (in the case of rivers), habitat for aquatic organisms, etc. The variables by which these characteristics are measured could conceivably reach a point where it has the potential to cause harm to the aquatic ecosystem.

Definition of a stressor

A stressor could be any substance or circumstance related to the aquatic environment, which could cause the aquatic ecosystem to lose sustainable ecological function. A pollutant would, by definition, be a stressor. The concept "pollutant" (in the definition of the NWA) is a subset of the concept "stressor". It should, however, be noted that a stressor may also include a set of variable values that individually would not necessarily have constituted a threat to human or aquatic life, but in combination could pose a threat. For example:

- Substances not in any way necessary for life, e.g. DDT, mercury and cadmium
- Substances necessary in the physiology of life in trace amounts (such as cobalt, zinc and copper) or in moderate amounts (such as salts and acids/alkalis) but which are either present in excess, or, chronically absent.
- Flow which is different (either higher or lower) from that which is natural to the time and place and to which organisms have become adapted over centuries.
- Modification of the in-stream habitat of organisms to a state where it is hostile to the organisms expected at the time and place.
- The presence of biota which are foreign to the time and place and which competes with indigenous biota.
- A critical combination of the first two above, which is manifested as a measurable toxic effect of unidentified origin such as estimated in whole effluent toxicity (WET).

Stressor diversity

Each of these stressors exists because they are deemed a possible cause of a specific effect (e.g. a loss of sustainability). Consequently, any of them could result in "loss of sustainability". The diversity among ecological stressors results from a diversity in:

- Temporal and spatial scale on which stressors have an influence.
- The units in which stressors are quantified.
- The end-points that are applied to the assessment of hazards related to each stressor.

Given that the ultimate guiding principles of water resource quality management are sustainability and equity, there is a need to compare these diverse stressors. The concept of risk is proposed a suitable basis on which stressors can be compared as well as managed.

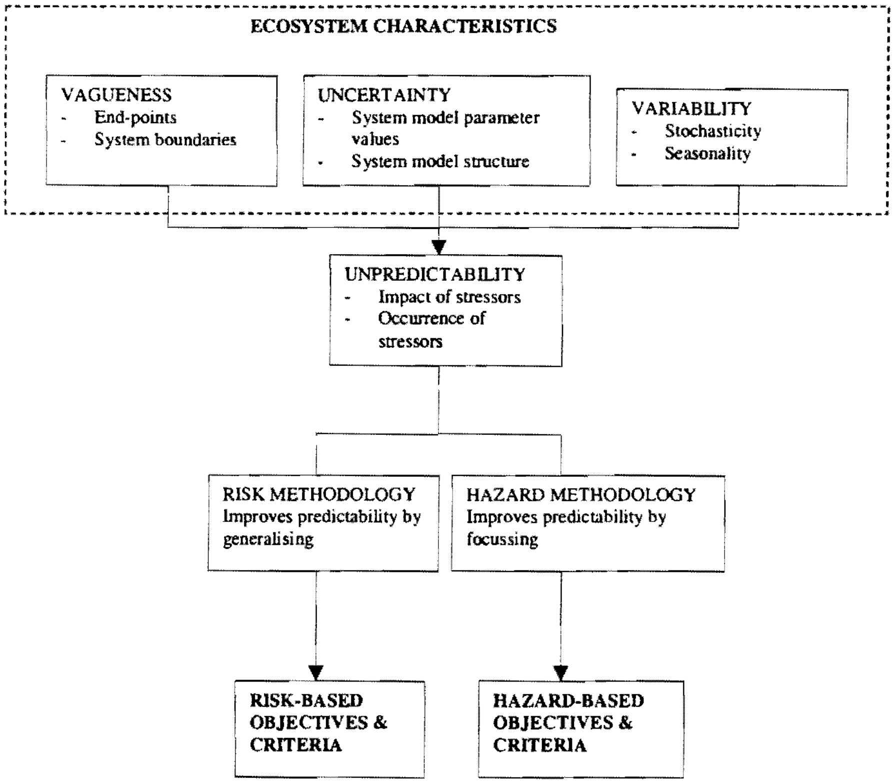

Figure 2
Ecological characteristics and their relationship with risk and hazard methodology

Ecosystem characteristics

A number of biologists consider ecosystems to be unpredictable or even chaotic in its behaviour (Grimm and Uchmanski, 1994). In terms of the NWA goals it is assumed that enough underlying order does exist to draw some conclusions on the response of a system to stimuli and to discount chaotic behaviour. There will still be some unpredictability and these are ascribed to three ecosystem characteristics: variability, uncertainty and vagueness (See Fig. 2).

Variability

Not only is variability commonly encountered, but organisms may be dependent on it. Hydrological conditions, seasonal cycles and variable response thresholds of individual organisms may all contribute to the survival of species. At a deterministic level, this variability may be seen as a source of unpredictability (See Fig.2)

Variability is recognised as a natural characteristic of biota (e.g. Brown, 1993; Grimm and Uchmanski, 1994; Kooijman, 1994). Several types of variability could be encountered. For example, there is a variability in individual response of the biota to a given stressor exposure (e.g. Hathway, 1984). The response variability can be represented by a cumulative response function, which expresses the cumulative fraction of the exposed population displaying a given level of response. This type of function would be analogous to the classic dose-response curve of toxicology, except that the shape of the curve need not necessarily be the same for all stressors. Although these functions may not necessarily be measurable in controlled laboratory experiments, a combination of field observation and expert interpretation is likely to provide an estimate of the stressor-response relationships. In this regard, the use of a Bayesian statistical approach rather than a strict frequentist approach may be indicated (Frey, 1993).

Spatial heterogeneity and stochasticity also impact on many processes in the aquatic environment, such as rainfall and sediment-solute-water interaction, which underlies the variability in the extent to which biota are exposed to stressors (O'Neill et al., 1979; Steinhorst, 1979; Crabtree, et al., 1987; Novotny, et al., 1994; Shine et al., 1995; Canale and Seo, 1996; Kapoor et al., 1997).

In the light of the ubiquity and necessity of variability in the ecosystem, it should not be viewed as a nuisance that can be ignored or even factored out by assumptions. Whichever approach is used in resource management should explicitly recognise this characteristic.

Uncertainty

Uncertainty in the sense used here is a characteristic of the human observer and stems from an imperfect knowledge of the system in point. A comparison between uncertainty and variability is presented in Table 2. Frey (1993) identifies two kinds of uncertainty: model uncertainty and parameter uncertainty.

The model uncertainty in the case of ecosystem models is due to the fact that with imperfect knowledge of a specific ecosystem's processes and mechanisms, there may be several conceptually valid options based on the study of other similar ecosystems or mechanistic models. There may, or may not be some means to weigh the model validity and, hence, the predictions made in this way may all be valid from the point of view of the observer. Only further measurement may reveal which of the models or combinations of models are truly valid. The stress responses may be quite precise, but the discrimination among the model choices

TABLE 2
Some of the characteristics of uncertainty and variability with particular reference to ecological models (based on Frey, 1993 and USEPA, 1997)

Characteristic	Uncertainty	Variability
Source	Lack of empirical knowledge of the observer or imperfect means of observation.	True heterogeneity inherent in a well-characterised population
Impacted by:	Model uncertainty • model structure • range of conceptual models Parameter uncertainty • random error due to imperfect measurement • systematic error (bias) • inherent stochasticity or chaos • lack of empirical basis • unverified correlation among uncertain quantities • expert disagreement on data interpretation	Individualism in response Lack of representative data Aggregation dimension (e.g. time or space)
Description	Probability distribution	Frequency distribution
Effect of more data	Reduces	Same but more precisely known
Applicability of standard statistical data analyses	Understated (due to focus on random error to the exclusion of bias introduced by variability)	Overstated (due to inclusion of measurement error)

may be blurred. This phenomenon is exacerbated by parameter uncertainty. Even when the specific model used to predict effects is known, very often the parameter values are wholly or partially unknown or the number of parameters are unknown. Some sources of parameter uncertainty are listed in Table 2.

These observations imply that in terms of ecologically oriented water resource management, it may be practically impossible to define a specific set of conditions that can be defined as representing "unsustainability". Sustainability will be a function of an uncertain array of possibly stochastic processes. Furthermore, the assessment of sustainability is dependent on a model which is uncertain to a greater or lesser degree and which is subject to variability. The exact point at which the system loses its sustainability can not be described deterministically, but rather in terms of the probability of reaching a condition of unsustainability.

A major problem in ecological goal-driven resource management is the uncertainty in the conceptual model relating the higher level concepts (such as sustainability) to lower level management varaibles (such as quantity and quality). It involves, *inter alia*, uncertainty in stressor-response relationships, uncertainty in the system boundaries and the interactions within the ecosystem (See Appendix 1). Deterministic answers are often not feasible or simply impossible and so decisions have to be based on uncertain information about a variable system. This emphasises the necessity for the use of probabilistic or possibilistic tools in water resource management to ensure protection of aquatic ecosystems.

Vagueness

This is also a characteristic of the human observer, but unlike variability and uncertainty as used above, it is not related to the content of one's knowledge, but to the state or type of one's knowledge. This may result, for example, when different lines of evidence in the assessment of sustainability contribute conflicting information. While this may superficially appear to cast serious doubt on the scientific tenability of the information, this phenomenon may simply result from different levels of assessment (e.g. different spatial and temporal levels, different levels of organisation, etc.). While the solution to this problem is outside the scope of this study, it is clear that a simple determinsistic approach will be inefficient and misleading.

Risk as a concept and an approach

In a colloquial sense, risk may refer to the gravity of the consequences when a mishap occurs or the potential that an undesired outcome may result from an action. The colloquial definition emphasises the hazard (or potential of causing an effect) resulting from an event while the latter definition emphasises the probability. In both cases there is a measure of dimensionality to risk; either the description of the hazard, or the specific consequences for which the probability is estimated.

Definition of risk

The concept of "risk" was defined in 1901 for the actuarial sciences as "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1990, p16) or the probability of observing a specified (undesired) effect as a result of a toxic chemical exposure (Bartell et al, 1992), or, simply, the possibility of suffering harm from a hazard (Haas, 1993). For the purpose of the reserve, a definition is favoured that is essentially dimensionless: Risk is the likelihood that a loss of sustainable ecological function will occur.

This definition emphasises two important aspects:

- An *a priori* decision as to what the undesired event is (i.e. loss of sustainable ecological function)
- A realisation that there is uncertainty about the event which is expressed in terms of a likelihood.

It may not be possible to assess the likelihood of this event directly ('statutory risk') and it may be that the risk of surrogate events may have to be assessed ('surrogate risk') in order to assess the statutory risk.

Hazards and risk

A hazard, in contrast to risk, refers to the potential that a situation has to cause harm. The hazard is not equivalent to the risk it entails. The hazard is a characteristic of the stressor that emphasises what could happen if the ecological entity is exposed to the stressor. It does not express how likely it is to happen since that depends on the situation being assessed.

For example: An endocrine-active substance is discharged to a river. It is known to cause testicular feminisation in fish at a level of 1 mg/ℓ. Its median lethal concentration for fish is about 600 mg/ℓ but its solubility in water is limited to 15 mg/ℓ. At the solubility limit it is unlikely to cause more than 10% mortality in a fish population. There are two hazards involved: mortality and population extinction through inhibition of fertility. If its concentration is managed to just below the solubility limit, the mortality risk is very low, but the population extinction risk is very high. In both cases there may be a hazard of unsustainability, but through different mechanisms. The risk will be determined by, for example, the occurrence of the substance as brief pulses followed by periods of very low concentrations, or, a fairly constant level between 1 and 15 mg/ℓ. It is conceivable that the risk in the first instance is lower than that in the second instance.

Expressions of likelihood

Likelihood is used in the definition of risk because there are sources of uncertainty and variability in both the effect and the exposure components of risk. Likelihood may be expressed in terms of:

- mathematical probability which is a product of probability theory, or
- mathematical possibility which a product of fuzzy logic.

Probability expression of likelihood

For an effect E (e.g. loss of sustainability) the probability that E is true is expressed as P(E). It is customarily assumed that P(E) will have a minimum value of 0 and a maximum value of 1.

P(E) may express either or both of two points of view:

- There is enough evidence to suggest that out of 100 repeated observation of E, in a 100*P(E)% of the observations E will be true, or
- There is enough evidence to make the observer believe that E will be true 100*P(E) % of the time.

The difference in interpretation is that in the first case the emphasis is on the frequency that E is true, while in the second case the emphasis is on the confidence induced by the body of evidence suggesting E to be true.

In many real ecological assessments there are not enough data from which a limiting frequency can be deduced from which P(E) can be inferred. However, there might be enough circumstantial or other indirect evidence that E might be true. P(E) would then express the confidence that E could be true.

Possibility expression of likelihood

A more serious problem than a lack of observations faces the assessment of ecological risk. The effect E might not be a clearly defined event. Loss of sustainability is a case in point. The loss of sustainability (or more precisely the point at which sustainability is lost) is not very clearly defined. This means that it not so easy to define E as being true or not. This calls for a multi-valued logic as opposed to a binary logic to express partial truth such as is found in fuzzy logic (Klir and Yuan , 1995). Possibility theory, which is based on fuzzy logic as opposed to probability theory, which is based on binary logic (Dubois and Prade, 1988) may serve well to express likelihood pertaining to the reserve. Such expression of likelihood in the context of the reserve was investigated by Jooste (2001 a).

Risk and hazard approaches

Resource management implicitly requires predictive ability for decision-making. It would not be sensible to suggest a change in a parameter value unless there is reason to believe that it will result in some advantageous effect.

In predicting or projecting an expected ecological effect there are two major aspects regarding stressors that need to be known: the way in which the target ecological entity reacts to changes in stressor level (i.e. stressor-response) and to what extent the target entity is exposed to the stressor. There are sources of unpredictability in both these aspects.

There are primarily two approaches to deal with ecological predictability problems (Fig. 2): the hazard approach and the risk approach. These approaches are both effect-based, but they differ in the way in which they deal with sources of unpredictability.

The **hazard approach** focuses the basis for decision-making by simplifying both the stressor-response and stressor occurrence by (necessary) assumptions. For example: the response variability, which is an inherent characteristic of the ecosystem, is simplified by selecting a stressor value that corresponds to an assumed "acceptable level of effect". This stressor value is then an assessment criterion value.

The criterion value is then interpreted to mean that all stimulus values less or equal to the criterion are acceptable, while all values above the criterion are unacceptable. The existence of a hazard is evaluated for each stressor value as it occurs.

Consequently, the hazard approach focuses both the stressor-response and -occurrence to single numbers, which are then compared.

The **risk approach** generalises the basis for decision-making

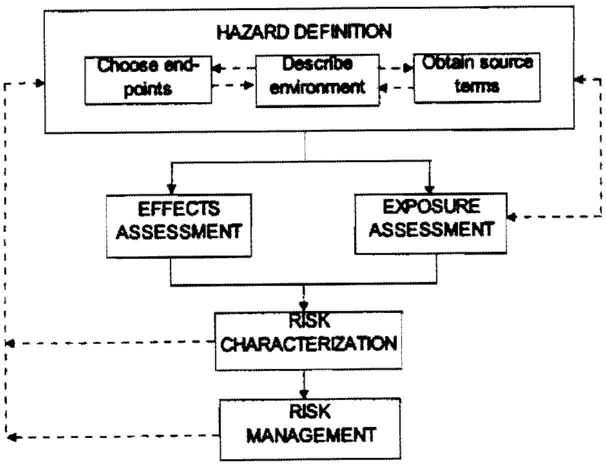

Figure 3
The basic elements of an ecological risk assessment where an ecological stressor and its source has been identified (Suter, 1993)

by incorporating as much of the relevant evidence as possible. It uses as much as is known about the relationship between stressor and response and about the occurrence of the stressor. It recognises that there may be a continuum of response over the stressor value domain at the point or in the area where an assessment is needed.

In the context of the resource management *vis-à-vis* the ecological reserve, where other uses have to be weighed against reserve goals, a risk approach might well be more flexible than a hazard approach.

Ecological risk assessment

Risk assessment is an array of techniques that is primarily concerned with the estimation of the likelihood and magnitudes of events. The likelihood element implies that in principle there is a continuum of risk from infinitely small (practically zero) to very high (practically certain). Due to practical limitations, coarser resolution (e.g. small, moderate, or high) is also used. It has become one of the most widely used techniques in environmental decision-making under uncertainty and has been the subject of intensive investigation by both the USEPA and the American National Research Council (NRC, 1994; USEPA, 1998). Protocols for both environmental and ecological risk assessments have been well-established.

Protocols for the assessment of ecological risk (ERA) have been produced by various organisations such as the USEPA. The basic elements of the ecological risk assessment process are outlined in Fig. 3 and discussed below. A generic adaptation of the USEPA protocol for South African environmental assessment and a more extensive discussion of the elements of an ERA have been produced by Murray and Claassen (1999).

There are a number of features of ERA that need to be considered in applying the methodology in water resource management:

- ERA can be performed at various levels of sophistication depending on the management need and the data input quality. The assessment ranges from qualitative through point estimates to full probabilistic assessments.
- The management goal under the NWA (and, therefore, the statutory end-point) for ERA is loss of sustainability. Assessing the satutory risk is usually difficult since it is unlikely that data will generally be available to assess the likely loss of sustainability in any given stressed aquatic ecosystem. It is more likely that data relating to lower level phenomena are available. A conceptual model (such as the example in **Appendix** 1) is required to project the uncertainty in loss of sustainability from knowledge of the measurable parameters. Such a projection model will relate the surrogate risk to the statutory risk.
- Each stressor risk can be assessed separately and aggregated later. Jooste (2000) and Jooste (2001) investigated a model for aggregating the risk for a number of diverse stressors.
- The ERA process explicitly makes provision for consultation with parties outside the management group. The NWA makes provision for public comment on the reserve. This affords the opportunity to consider a variety of opinions on the reserve. The ERA process also allows for consideration of specific values outside of the scientific opinion inherent in the process.

Discussion

A hazard-based precautionary approach might be administratively ideal. A pragmatic version of a hazard approach was suggested by Van der Merwe and Grobler (1990) by using the pollution prevention approach for hazardous chemicals and the receiving water quality objectives (RWQO) approach for the non-hazardous substances. In terms of the ecological reserve, the distinction between hazardous and non-hazardous is difficult and the aggregation of diverse stressors is not possible with RWQOs. In addition, using hazard-based RWQOs (e.g. those based on the South African Water Quality Guidelines (SAWQG, 1997)) does not allow for effect-based management as implicitly required under the NWA. While the principle of using in-stream objectives is sound, greater benefit would derive from using risk-based objectives (See **Appendix** 2).

The implication of the NWA, stressor diversity and the characteristics of the ecosystem allow for the use of an ecological risk approach because of its formulation in terms of likelihood. In particular, it is noted that:

- The NWA requires sustainable use. This implies that use of the resource needs to be balanced against its protection. A hazard approach to water resource management tends to be inflexible when use is permitted (or even encouraged). This is because only some of the stressor effect information and some of the stressor occurrence information are used to assess resource status. On the other hand, a risk approach allows more of both effect and occurrence data to be used.
- The diversity of stressors that impact on the aquatic ecosystem cannot be handled in an integrated fashion by a hazard approach. Commonly, a hazard will be defined in terms of stressor measuring units such as concentration, flow rate, etc. A hazard approach does not inherently allow for ranking stressors or managing for combined effect. A risk approach has the advantage of placing stressors on a common, practically unitless basis.
- The characteristics of the ecosystem and our knowledge of it such as the necessity of variability and the epistemic uncertainty mitigates against making any information regarding the system and its response to stressors redundant. Such redundancy is

necessarily a part of the hazard approach to resource management. The risk approach, by contrast, tends to be less wasteful of available data.

The use of risk does not preclude a precautionary approach. Precaution is introduced by, for example, conservative assumptions or policies regarding:

- Risk acceptability criteria (what levels of risk are acceptable for each class)
- Acceptability of stressor-effect data (e.g. rejecting data that suggest questionably high tolerance)
- Stressor occurrence estimation (e.g. not accepting stressor degradation for conservative substances)

Although risk assessment may yield continuous assessments, setting risk acceptability criteria could generate dichotomous assessments. Such criteria may comprise of:

- a *de minimis* risk criterion, i.e. a criterion that indicates that the risk is too small to be of any concern and the situation that gives rise to it does not need serious attention, and
- a *de manifestis* risk criterion, i.e. a risk that is unacceptably large and the situation that gives rise to it, one that is unacceptable.

In the present context, where risk is descriptive of a viewpoint of an observer, both *de minimis* and *de manifestis* risk are more likely to be generated in the water resource management policy domain than in a strictly scientific domain. The range between the *de minimis* risk value and the *de manifestis* risk value can be divided into an arbitrary number of values to correspond with the resource classification required under the NWA. These would then give rise to resource risk objectives (RROs).

The RROs would then reflect the aggregate risk of all stressors in the resource (as defined in the definition of the reserve). These RROs could then be used to derive site-specific resource quality objectives that take cognisance of the local surrogate risk parameters as well as the characteristics of the known stressor sources in a catchment. An example of this is given in Jooste (in press).

Conclusions

Ecological risk could serve as a useful approach in certain aspects of water resource management. Interpreting resource classification, as required in the NWA, on a risk base, will assist in deriving resource quality objectives that are both efficacious and flexible.

An ecological risk approach is not a panacea for water resource management. It requires consideration of the scientific data and its relation to human values. It reduces decisions from a purely mechanical process to one that requires explicit action. While this may be difficult in some situations, it increases the flexibility and transparency of the catchment management process while simultaneously assuring that the goal of protection of the ecosystem is attained to the extent possible.

Risk as a tool, although not exclusively dedicated to, is best applied in a risk management framework. In such a framework the objective of risk based decision-making would be to balance the degree of risk to be permitted against the cost of risk reduction (not necessarily only in monetary terms) or against competing risks.

- Formulating a policy for the use of risk-based methods which should serve both to guide the development of an ecological risk assessment ethic in South Africa (e.g. it would address the perception that using risk is merely an excuse for doing nothing (Tal, 1997)).
- Developing a framework for risk-based resource quality management and synthesising this with the current institutional framework.
- Defining and evaluating an acceptable risk range bounded by the *de manifestis* and *de minimis* risks.
- Discretising the acceptable risk range in keeping with the classification of water resources and formulating realistic risk-based objectives in keeping with the ecological reserve.
- Investigating methodologies from the information sciences by which the scarce data and expert knowledge can be brought together to produce the information, particularly the stressor response information, needed to calculate the stressor specific risk.

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Appendix 1
A conceptual model for end-point projection

It is unlikely that data will generally be available to assess the likely loss of sustainability in any given stressed aquatic ecosystem. It is more likely that data relating to lower level phenomena are available. A conceptual model is required to project the uncertainty in loss of sustainability from knowledge of the measurable parameters. A phenomenological inference model for the ecological reserve with a precautionary approach may be based on the following postulates:

- The reference state for the model is the pristine system. The pristine system has all the characteristics (including the potential for sustainable use) that could be wished for. It is assumed that the reference state's only fixed characteristic is its 'degree of correspondence to the pristine state', but that the values of the descriptors used to characterise this state would be spatially and temporally variable.
- For a system that is managed to be under constant stress (as most South African surface water systems are due, to the semi-arid nature of most of the country), integrity (and by implication resilience) is lost more easily than in a comparable system subject to infrequent high intensity stress (Rapport et al., 1995). This means that both acute (in the sense of high-level short-duration) stress, and chronic (in the sense of low-level long-duration) stress should be addressed in resource management.
- It is provisionally assumed that a specific point exists where the sustainability of the system is lost (the system 'crashes' with respect to sustainable use). This point is generally unknown, but the likelihood of approaching this point can be assessed on a "grey scale". The uncertainty in describing this point is similar in the uncertainty in the critical level of loss of integrity that corresponds to this point. The state of integrity of the system is determined by its state of biodiversity and the deviation from the natural temporal and spatial patterns of flow and water chemistry.

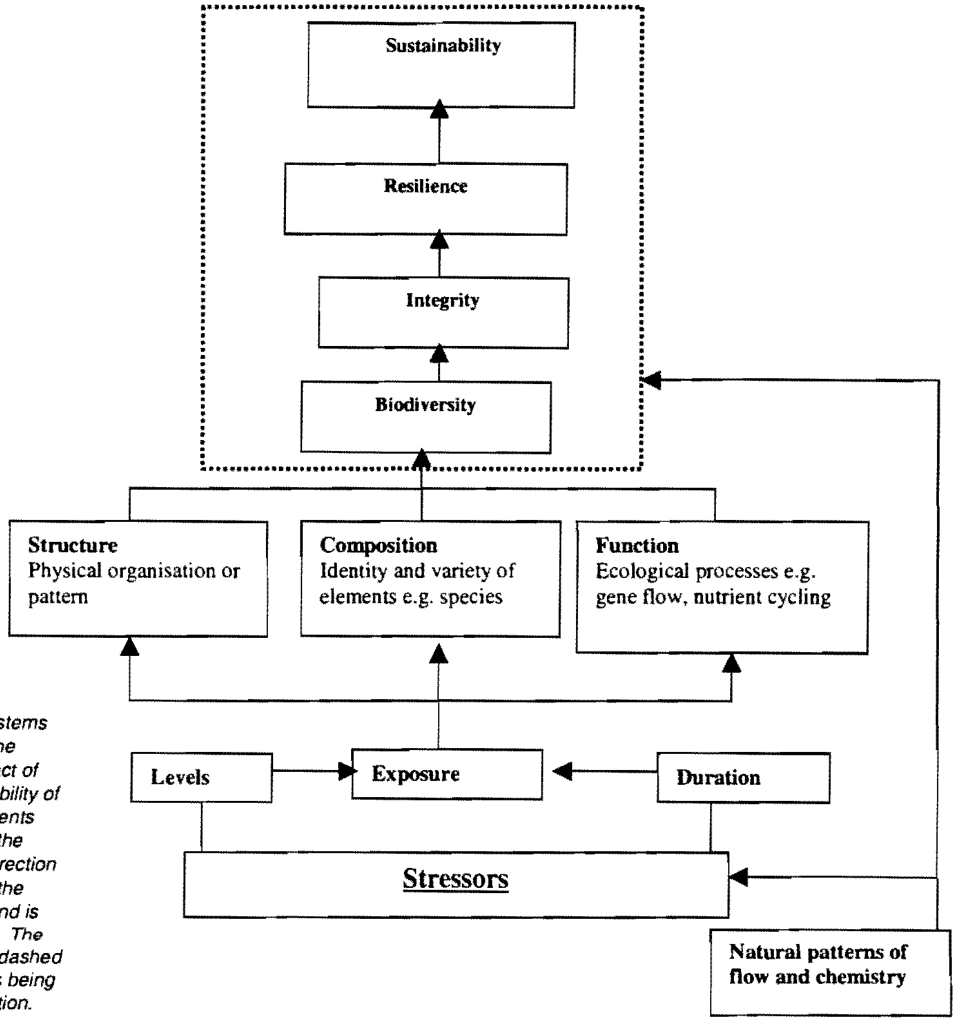

Figure A1
A phenomenological systems model for inferring the uncertainty of the impact of stressors on the sustainability of the system. The elements should be read as : "the uncertainty in...". The direction of the arrows shows the direction of influence and is interpreted as "affects". The elements within the thick dashed line may be combined as being equivalent by assumption.

- Biodiversity is dependent on the composition, structure and function of the system (each at several levels of organization from molecular to landscape level) in relation to what it could have been in an undisturbed, pristine system. Biodiversity as a variable indicating stress is subject to an interpretation of the individual importance of species. Redundancy is possible or even probable in an ecosystem and the real question is how much redundancy could be lost without pushing the system to the edge of some irreversible, catastrophic change (DeLeo and Levin, 1997). The conservative assumption would be that all species are equally important and that loss of species systematically undermines integrity.
- A further precautionary assumption is that the system under consideration is isolated and repopulating from refugiae outside the borders of the system is impossible.

A conceptual phenomenological model based on these postulates is presented in Fig. A1. In this model the arrows indicate how the uncertainty in one variable affects the uncertainty in another. The elements within the thick dashed line are assumed to be logically equivalent in the sense that the epistemological uncertainty in the impact of one on the other is similar. This assumption need of course not hold if more specific information is available.

Each of the propositions regarding impact (represented by the arrows in Fig. A1) of this conceptual model is based on a sense of expectation founded on the assessor's knowledge base, experience and perception of the specific situation being assessed.

Logically, the certainty in a higher level variable cannot be higher than that of a lower level variable. This means that there is a greater uncertainty in the statutory risk than in the surrogate risk. This model helps the assessor to select an end-point and the same time to describe the uncertainty in the risk assessment goal.

Appendix 2
A risk interpretation of the current SAWQG criteria

Suppose a specific effect gives rise to an event E in an ecosystem that is subject to n different stressors. In general, each different stressor i will give rise to E_i. The combined probability of effect is given by (DeFinetti, 1990):

$$P(E) = P\left(\bigcup_{i=1}^{n} E_i\right) = \sum_i P(E_i) - \sum_{i,j} P(E_i E_j) + \sum_{i,j,h} P(E_i E_j E_h) - \ldots \pm P(E_1 E_2 \ldots E_n) \quad \text{(A1)}$$

where $P(AB)$ denotes the probability of the conjunction of A and B. The form of $P(AB)$ depends on the independence of A and B. In the case where the occurrence of A is logically independent of B, then $P(AB)$ is expressed as $P(A)P(B)$. The resulting boundaries on the effect probability is given by Eqs. (A2).

$$\max_i\{P(E_i)\} < P(E) \leq \sum_i P(E_i) \quad \text{(A2)}$$

A safety factor γ_i where ($\gamma_i \geq 1$) applied to a risk is a_i for stressor i, to accommodate uncertainty of some kind, then the implied risk b_i for stressor i is: $b_i = a_i/\gamma_i$. If the individual stressor risks are assumed to be logically independent, then, from Eq. (A1), the total risk can be expressed as Eq. (A3).

$$P(E) = \sum_i \gamma_i b_i - \sum_{i,j} \gamma_i \gamma_j b_i b_j + \ldots < \sum_j \gamma_i b_i \quad \text{(A3)}$$

Comparing the situations where there are n different stressors present to the one where there are m different stressors:

$$\frac{P(E)_n}{P(E)_m} < \frac{\sum_{i=1}^{n} \gamma_i b_i}{\sum_{i=1}^{m} \gamma_i b_i} \quad \text{(A4)}$$

If $m > n$ then the right-hand side of Eq. (A4) is less than one if γ_i is constant. This implies that if a constant safety factor is used in the derivation of criteria, the total risk to the ecosystem increases as the number of (potentially) additive stressors increase. Alternatively, if a constant total risk is assumed (which should be independent of the number of stressors) then the risk ratio should be 1 and, therefore, Eq. (A4) becomes Eq. (A5):

$$\sum_{i=1}^{m} {}^m\gamma_i {}^m b_i < \sum_{i=1}^{n} {}^n\gamma_i {}^n b_i \quad \text{(A5)}$$

If the safety factor is to be independent of the stressor and the individual stressor risk levels are constant then ${}^m\gamma > {}^n\gamma$, which means that the safety factor is dependent on the number of stressors if the total risk is to be kept constant.

In the derivation of the current SAWQG criteria provision is made for a target water quality range (TWQR, abbreviated to T), a chronic effect value (CEV, abbreviated to C) and an acute effect value (AEV, abbreviated to A) (Roux, *et. al.*, 1996; SAWQG, 1997). Although risk is not the explicit basis for derivation, each of these implictly represent a risk a_i, c_i and t_i respectively. By definition $c_i > t_i$, but there is no way of comparing a_i and c_i directly since they refer to different end-points.

There is an implicit maximum total acceptable risk of effect E of $max\{a_i, c_i\}$ for any single substance i. If the management goal is that the substance concentrations are lower than the criterion values, then from Eq.(A2) the total risk, $P(E)$, will be expressed as in Eq (A6).

$$P(E_A) \leq \sum_{i=1}^{n} a_i \qquad P(E_C) \leq \sum_{i=1}^{n} c_i \quad \text{(A6)}$$

If all the stressors acted *independently* then, in which case the implicit risk condition is met. However, if stressors k and l, for example, interact with the target organisms by some common mode of action, so that their effect is additive in some way (Calamari and Vighi, 1992), then the probability of their combined effect can be expressed in terms of the joint probability, say $P(E_A/A_k, A_l)$ which, according to Eq. (A3), will always be larger than $max\{a_k, a_l\}$.

This means that if:

- There is any additivity of effect among the stressors present and management up to the criterion levels allowed for each stressor, then the probability of combined effect will be larger than the implied maximum acceptable effect probability. Consequently, management of stressor levels up to the criterion values will logically result in an "unacceptable" level of effect.
- Safety factors had been applied in the derivation of the criteria (Kooijman, 1987), so that the actual risk implied by the criteria is less than the acceptable risk, then the margin of safety afforded by these safety factors depends on the number of stressors assumed to be present (Eq. (A5)). Chapman et al., (1998) point out that current application of safety factors is largely a matter of policy and not of empirical science and that injudicious use may result in useless overprotection.

A model to estimate the total ecological risk in the management of water resources subject to multiple stressors

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Abstract

The disjunctive convolution of independent individual stressor risk is presented as a model to estimate the total expectation of ecological effect for a water resource, subject to several different and metrically disparate stressors. This method makes use of the exposure and effect assessment data of the risk assessment procedure for each individual stressor given that the end-point is the same. A hypothetical case study illustrates how total risk could be used as an ecological goal-oriented tool in catchment management.

Glossary

ERA:	Ecological risk assessment
Hazardous:	Having the potential to cause an (undesired) effect
Stressor:	An anthropogenic substance, form of energy or circumstance that may cause a change in ecosystem integrity
N(x,y) :	The normal (Gaussian) distribution with median x and standard deviation y
LN(x,y):	The log-normal distribution with median x and standard deviation y
Weibull(α, β):	The Weibull distribution with scale parameter and location parameter
[a, b] :	The interval from a to b where both a and b are included
(a, b) :	The same interval with both a and b excluded.

Introduction

The management of a water resource with a specific ecological goal in view can be particularly problematic when the water resource is subject to multiple diverse stressors such as chemical substances, deviations from expected flow, habitat degradation etc. An example of this is found in the South African National Water Act (Act 36 of 1998). It makes provision for an ecological Reserve, a quantity and quality of water to (*inter alia*) protect aquatic ecosystems in order to secure ecologically sustainable development and use of the water resource. The provisions of the Act pertain not only to the regulation of discharges to surface water but also to abstraction from the water resource as well as to the quality of the instream and riparian habitat necessary for assuring the protection of the aquatic ecosystem. At the same time, it is recognised that South Africa is a semi-arid country (DWAF, 1986) and consequently a fine balance is needed in water resource management between protection and utilisation. Here the ecological goal of sustainability must be achieved in aquatic ecosystems subject to diverse stressors such as discharge of substances, the abstraction of water and the destruction of the physical habitat which occur to a greater or lesser degree.

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It has been suggested (Jooste and Claassen, submitted to *Water SA*) that a probabilistic effect-based approach has some potential for application to the problem of multiple stressor impacted water resources. A method is suggested whereby an adaptation of the conventional ecological risk assessment methodology can be used to assess the overall risk of multiple stressors in the management of catchments with a view to maintenance of the ecological Reserve.

The problem of a multiple stressor environment

One of the difficulties of ecological water resource management in a multiple stressor environment is the problem of predicting the integrated effect of co-occurring stressors of different types. The disparity among stressor measures necessitates the separate consideration of stressors and their effects. The stressors are then regulated, assessed and controlled separately. At the same time, these stressors may add to a disruptive effect. The integration of effects has been attempted mechanistically on a physiological basis by considering the production of stress proteins (originally referred to as heat shock proteins). These are grouped into three classes:

- those related to the heat shock phenomenon;
- glucose regulated proteins; and
- stressor specific proteins such as metallothionein (Di Giulio et al., 1995; Shugart, 1996).

The stress protein response becomes an integrated signal for environmental stress. While such a mechanistic approach is likely to produce more accurate assessments, its data requirements are extensive. At a more phenomenological level, it may be possible to estimate the probability of stress-induced changes by considering the probability of separate stress events.

Some observations regarding the aquatic ecosystem

The ecological status of a resource is determined by the dynamics and kinetics of interactions of aquatic animals, plants and processes that determine the function, composition and diversity that characterise the ecosystem. Water resource management objectives and their associated criteria must reflect the following inherent ecosystem characteristics if they are to achieve their goal:

A variety of stressors (e.g. habitat, water quality, and flow (Quinn and Hickey, 1994; Armitage and Gunn, 1996; Schofield and Davies, 1996; Dyer et al., 1998)) may be at work at various spatial and temporal scales and yet result in the same unacceptable effect. For example, a fish species may disappear from a river either because of severe chemical contamination, over-harvesting of the species, impairment of crucial breeding habitat or simply because there is no water in the river.

There is an innate and irreducible inter- and intraspecific variability in biotic response to a given stressor. Biotic systems are characterised by variability (O'Niell et al., 1980; Kooijman, 1987; Brown, 1993). The variability observed in the response of organisms may derive from an underlying stochasticity in individual susceptibility (Mancini, 1983; Breck, 1988). There is also an underlying stochasticity in aquatic environmental interactions which produces temporal and spatial variability in stressor levels.

There are limits to the scientific certainties about any given natural biotic system which impact, *inter alia*, on the certainty of cause-effect relationships in the particular system. Uncertainty is largely a characteristic of the observer and his deductive processes. Since modelling, whether conceptual or mathematical, often forms a part of the deductive process, uncertainty may derive from:

uncertainty in future input to the model;
uncertainty in model structure and parameters; and
uncertainty in the application and validity range of the model and may well be reducible on presentation of more or better information.

The impact of uncertainty is so severe that the use of quantitative (usually deterministic) predictive models is disparaged by some biologists (e.g. Fryer, 1987). According to Holling (1996), there is "an inherent unknowability, as well as unpredictability, concerning the ecosystems and the societies with which they are linked".

In many natural ecosystems there is a dearth of detailed data about structure, function and composition (e.g. Cairns, 1986; Landers et al., 1988; Munkittrick and McCarty, 1995). Ecological knowledge is often descriptive rather than quantitative. Responses of organisms to stressors are normally continuous and discontinuities are normally an artifact of the resolution of observation. If the test population is large enough or the observation method discerning enough, the response of the population is essentially continuous (e.g. Hewlett and Plackett, 1952; Hathway, 1984)

The above argue strongly for a non-deterministic approach to the impact assessment related to, and management for, ecological goals. Jooste and Claassen (submitted to *Water SA*) suggested the application of ecological risk concepts to resource management in the context of the ecological reserve. The ERA methodology needs to be adapted to assess the overall risk.

Risk assessment

"Risk" has been defined as "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1990) or the probability of observing a specified (undesired) effect as a result of a toxic chemical exposure (Bartell et al., 1992). Risk has three necessary components: probability, target and effect; all of which require explicit statement.

"Risk assessment" is an array of techniques that is primarily

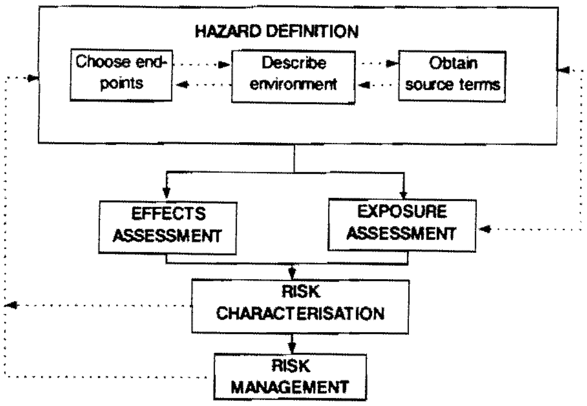

Figure 1
A diagrammatic representation of the predictive use of ecological risk assessment (from Suter, 1993). The dashed lines indicate feedback loops.

concerned with the estimation of the probabilities and magnitudes of events. ERA concerns itself with the estimation of the probability of specific ecological events taking place. These events could comprise a specific effect experienced by a specified target organism (or other ecological entity) when exposed to a stressor. A simplified outline of the procedure is shown in Fig. 1. An important feature is the choice of end-point which implies both target organism (or ecological entity) and level of impact (EPA, 1997a).

The ERA procedure described here is performed at different levels of sophistication (EPA, 1998). The effect assessment is sometimes reduced to generating a number, which, in the estimation of the assessor or the risk manager, represents an acceptable level of effect expressed in terms of a measurement variable such as the concentration of a substance in the water column. This concentration is known under different guises, depending on how it was derived, but is here called the acceptable effect concentration (AEC).

The exposure assessment feature derives a number, which is assumed to represent a suitable exposure scenario (e.g. the worst case exposed organism, reasonable worst case exposure, median exposure etc.), also expressed as a concentration. This is the exposure concentration (EC). Depending on the situation, the EC may either be predicted or measured. In its simplest form, i.e. a screening level risk assessment, the risk characterisation step involves the convolution of the effect level and the exposure level in the form of a ratio. The risk number is calculated as the ratio (DEPA, 1995): $R = AEC/EC$. At a screening level, it is only necessary to establish broad categories for this ratio. For example if $R \in [0,1)$ then no further calculation may be necessary; if $R \in [5, \infty)$ then the risk is assumed to be too high and other steps need to be taken to address the situation, while if $R \in [1,5)$ a more detailed risk calculation is needed. At more advanced levels the uncertainty and variability pertaining to the system and its models are brought into the calculation, yielding a probabilistic risk assessment.

The characteristics noted above, of the systems that are to be protected by the implementation of the ecological reserve, make the use of risk-based techniques such as ERA attractive. In an appraisal of the risk assessment and risk management in regulatory programmes, the Commission for Risk Assessment and Risk Management (CRARM, 1996) came to the conclusion "that it was time to modify the traditional approaches to assessing and reducing risks that have relied on a chemical-by-chemical, medium-by-medium, risk-by-risk strategy" and to focus rather on the overall goal of risk reduction and improved health status. They maintain that risk assessment was developed because scientists were required to go beyond scientific observation to answer social questions about what was safe.

Risk convolution

Each stressor acting on an ecosystem produces an individual risk or probability of effect. Each of these individual stressor risks can be estimated by ERA. In order to assess the expectation of all the stressors acting at the same time, the individual stressor ERA outcomes need to be convoluted. There are several mathematical operators that can be used to convolute stressor risk to reflect the total risk, including: maximum, sum and conjunction. In order to explore the use of each of these, it is necessary to formalise the description of the ecological objectives in probabilistic terms.

An ecological objective can be described in terms of events, with an "event" consisting of the information triplet {object, end-point, level}. For example, the information that "more than a 5% decrease in the expected biodiversity may cause an irreversible change in this ecosystem" gives rise to the objective: "the decrease in biodiversity should be less than 5%". This can be encapsulated in the event E = *{biodiversity, decrease, 0.05}*.

The event E can further be partitioned into events (DeFinetti, 1990) that relate to the various types of anthropogenic stress, such as toxicity (t), flow regime disturbances (q) and habitat degradation (h). Therefore, $E = E_t \vee E_q \vee E_h$ where E_t = *{expected number of species, toxic stress effect, 0.05}*, E_q = *{expected number of species, flow regime disruption stress effect, 0.05}* and E_h = *{expected number of species, habitat degradation stress effect, 0.05}*.

The total ecological risk is expressed by $P(E)$, which is the probability of the conjunction of the partitioned events, and therefore:

$$P(E) = P(E_t \vee E_q \vee E_h) \tag{1}$$

As a general case, suppose an event E involves a specific level of effect (specified by the assessor or risk manager) in an ecosystem subject to n different stressors. Therefore, each stressor i will give rise to E_i. The combined probability of effect (in set theoretical terms) is given by (DeFinetti, 1990):

$$P(E) = P\left(\bigcup_{i=1}^{n} E_i\right) = \sum_i P(E_i) - \sum_{i,j} P(E_iE_j) + \sum_{i,j,h} P(E_iE_jE_h) - \ldots \pm P(E_1E_2 \ldots E_n) \tag{2}$$

If E_t, E_q and E_h are all logically independent, then probability of the conjunction of individual ecological effects reduces to the product of the individual effect probabilities, and hence the application of Eq. (2) to Eq. (1) yields Eq. (3):

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)] \tag{3}$$

It is recognised that $P(E_t)$, $P(E_q)$ and $P(E_h)$ are joint probabilities of effect ε_x and exposure x so that: $P(E_x) = P(\varepsilon_x, x) = P(\varepsilon_x | x)P(x)$, where $x \in \{t, q, h\}$.

A distinction is made between logical dependence and causal dependence (Jaynes, 1996). Two events A and B are logically dependent if, for example, the occurrence of A implies the occurrence of B. This is different from the proposition "A causes B". If a reduction in biodiversity due to toxicity is inferred from the information at hand, then there is no possibility of inferring that reduction of biodiversity due to habitat stress will occur. This should not be confused with the situation where, for example, data at hand indicate that the probability of mortality due to toxic stress in conjunction with habitat stress is greater than that predicted by Eqs. (2) or (3). $P(E_x)$ should not be confused with $P(\varepsilon_x)$ (see below).

$P(\varepsilon_x | x)$ is defined as the probability of an effect given the event that stressor X is present at level x. This information is derived from a probabilistic stressor response relationship, which predicts the probability of a specified effect (of the same type as in the original n-tuple definition; i.e. the expected number of species in this case) as a function of exposure to a stressor. This implies that the value of $P(E_x)$ can simply be estimated from a probabilistic stressor response relationship and the probability of occurrence of exposure to a stressor x. Stressor response relationships are often evaluated empirically, although it might be necessary to partition each of the events in Eq. (1) into component events in order to get to a level at which sufficient empirical data can be collected to evaluate the event probability.

Furthermore, the effects ε_x may not be functions of one stressor only. It may be necessary to partition the event "existence of stressor X" into events that signify the occurrence of stressors that collectively manifest as stressor X: i.e. X is partitioned into occurrence of stressors $(X_1, X_2, \ldots X_n)$, where there are n stressors that make up the class of stressor X. Due to interactions among stressors, it may be necessary to evaluate $P(\varepsilon_x | X)$ where all n different stressors are present at the same time. Most often this will not be possible experimentally (except perhaps in the case of toxic stress), so that simplifying assumptions will have to be made. However if events X_i are logically independent then this reduces to (DeFinetti, 1990):

$$P(\varepsilon_x | X) = \sum_j (PX_j) \cdot P(\varepsilon_x | X_j) \tag{4}$$

It might be, that although the stressor occurrences X_i and X_j are independent, the effect ε is dependent on the co-occurrence of X_i and X_j. This might be due to some mechanistic interdependence such as synergism or antagonism in which case the occurrence of (X_i, X_j) might manifest as a new stressor Y. In this case $P(\varepsilon | X_iX_j)$ would be given by $P(\varepsilon_Y | Y) = P(\varepsilon, Y) | P(Y)$. Therefore, $P(\varepsilon, X_i, X_j) = P(X_i)P(X_j)P(\varepsilon | Y)$, where the value for $P(\varepsilon | Y)$ has to be evaluated experimentally. However, cases of true synergism among toxics, for example, are reported to be rare (Calamari and Vighi, 1992). The occurrence of synergism among other stressors may be possible.

A hypothetical case study

In an ERA for a stretch of river it was agreed between the risk manager and the risk assessor that the sustainability of the aquatic ecosystem can be expressed in terms of the end-point "a 5% decrease in biodiversity". Furthermore, three sources of stress (i.e. the hazards) were isolated:

Stressor 1 is the modification of the streambed and riparian zone resulting in destruction of habitat (independent of flow). This is reflected in habitat degradation which is expressed (hypothetically) as a percentage, where zero indicates no degradation and 100 denotes complete degradation. In the assessment, it is found that there are practically pristine sections as well as degraded areas in the river reach, so that the habitat degradation can be described by a normal distribution (see Table 1). It is proposed that the response of the system to habitat degradation (all else being equal) can be described by a Weibull distribution (Fig. 2a).

TABLE 1
Stressor Magnitude and System Response Modelling Functions

Stressor	Stressor response function P(E|x)	Stressor magnitude distribution P(x)
Habitat	Weibull(5, 50)	N(25, 7)
Flow	1-Weibull(15, 7)	LN(12, 1.3)
Toxics (Scenario 1)	Weibull(3, 2.715)	LN(3.8, 1.25)
Toxics (Scenario 2)	Weibull(3, 2.715)	LN(1.9, 1.25)
Toxics (Scenario 3)	Weibull(3, 2.715)	LN(0.95, 1.25)
Toxics (Scenario 4)	Weibull(3, 2.715)	LN(0.475, 1.25)

Stressor 2 is the water depth in the river. This is assumed to be directly proportional to the flow which is log-normally distributed for the reach under investigation. It is accepted by the river ecologists on the risk assessment team that the response of the system to this measure can be described by an adapted Weibull function as shown in Table 1 and Fig. 2b.

Stressor 3 is the presence of toxic substances in the river. These substances are unidentified and were established by whole effluent toxicity testing at the source discharge to the river. The level of these substances is expressed in terms of toxic units. For this situation a toxic unit has been defined as: 100/LC5, where LC5 is the 5th percentile of the mortality distribution for the test organisms with the concentration expressed as a percentage (DEPA, 1995). The toxic units were found to be log-normally distributed. From ecotoxicological studies, the system response to these toxics is approximated by a Weibull function (Fig. 2c).

It is assumed that the flow regime as described will not result in further habitat degradation by inducing changes in channel morphology. There has been no evidence to suggest an interdependency among the stressor effects. Consequently, the occurrence of effects resulting from these stressors is logically independent by default assumption.

Total risk calculation

The convolution expressed in Eq. (3) was used. The stressor-response profile is expressed as the probability of "a significant ecological effect" in the river reach and the result is expressed as the cumulative probability of effect ($P(\varepsilon_x | X)$. This type of result may be obtained from a site-specific study, expert opinion or system simulation modelling.

The stressor-specific probability of effect is calculated from the product of the stressor probability density and the probability of effect to give the probability density of effect for this river reach for each stressor X (stressor risk $p(E_x)$).

Since these stressors have been assumed to occur independently, Eqs. (3) and (4) were solved iteratively by randomly selecting the stressor risks from their respective density profiles to obtain the risk distribution for these specific conditions in this river reach. The random stressor magnitudes were calculated as described in Frey and Rhodes (1999). One thousand random samples were selected for each stressor. The stressor profiles, and conditional response probabilities are shown in Figs. 2a, b and c. The calculated risk distributions are shown in Fig. 3.

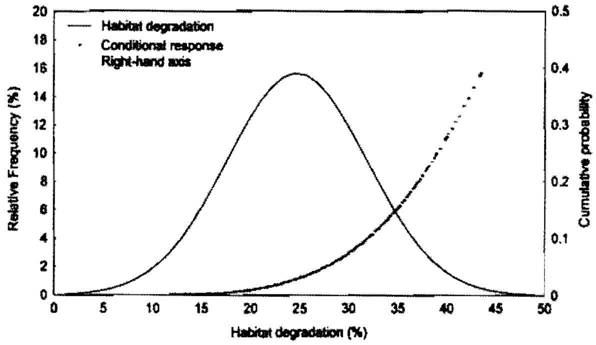

Figure 2a
Habitat degradation distribution as used in the Monte Carlo simulation and the conditional probability of system response (points referring to the right-hand ordinate scale)

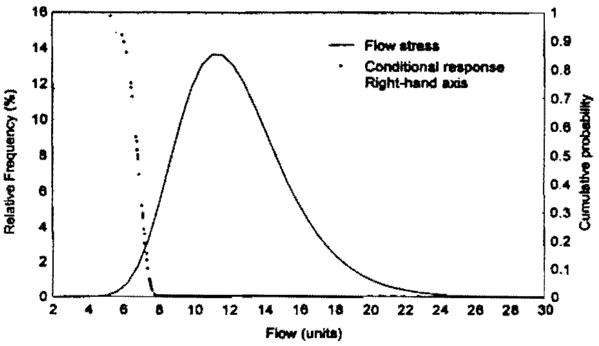

Figure 2b
The flow-related stressor magnitude distribution (solid line) and the corresponding conditional system response probability (point referring to the right-hand ordinate scale)

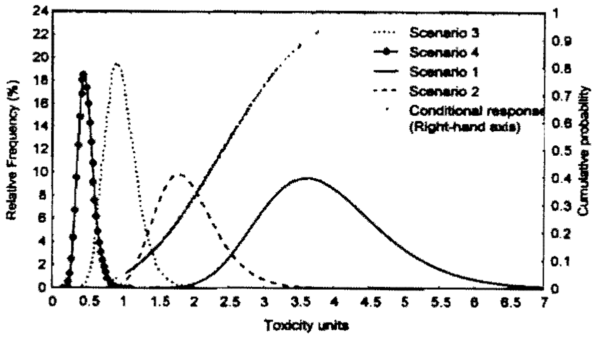

Figure 2c
The toxic unit distribution for the four scenarios described in the text (lines referring to the left-hand ordinate scale) and the conditional system response profile for the toxic substances (the points referring to the right-hand ordinate scale)

Risk ranking

The contribution of each stressor to the risk expectation for a river reach may vary depending on the stressor-response profile and stressor-probability profile. The conjunctive convolution model (Eq. (2)) predicts that, depending on the risk level allowed, differ-

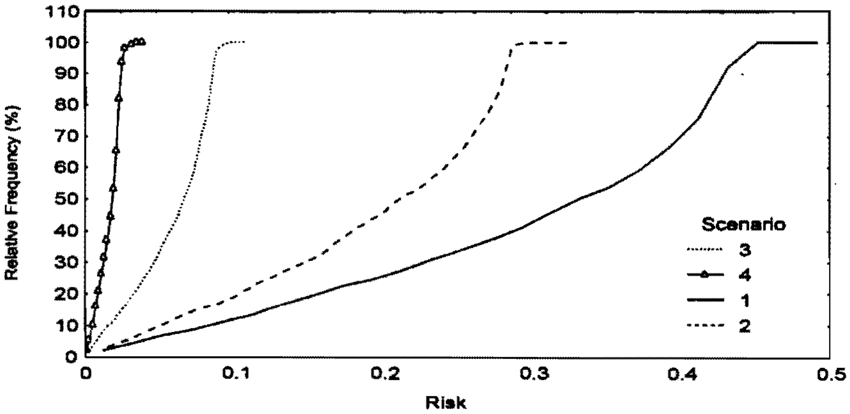

Figure 3
The cumulative probability profiles of the overall risk from the Monte Carlo simulation of the scenarios described in the text

ent stressors could dominate the overall risk in a catchment. It is possible to rank the risks, rather than the hazards, in a catchment and focus on those. In the example above, it can be seen from the stressor profiles, that the presence of toxics appears to dominate the risk contributions. The management objectives for stressors giving rise to lower risks could be set at levels in some way representative of the lower risks (e.g. median lower risk, i.e. median stressor risk excluding the dominant stressor risk). The sub-dominant stressors in the catchment need only be monitored (e.g. by means of the stressor probability profile) until the dominant stressor had been addressed. Periodic recalculation of stressor risks will reveal either the appearance of a new dominant stressor or the overall acceptability of the integrated risk.

The ratio of the individual stressor risks to the total risk is depicted in Fig. 4. It is apparent that in Scenario 1 (Table 1) above, the toxicity in the river is the major contributor to overall risk.

This can also be seen by inspecting the position of the response curve in relation to the stressor magnitude profile in Fig. 2c. Based on this assessment, it would seem likely that the relatively high overall risk (90th percentile of about 0.44) can be ameliorated by managing the system to a lower toxic unit level. For Scenario 2, the toxic unit median is set to 1.9. The corresponding overall risk 90th percentile is now less than 0.3 but still too high. For Scenario 3, the toxic unit median is adjusted to 0.95 and for Scenario 4 the toxic unit median is adjusted to 0.475. The individual risk ratio's for Scenario 4 is shown in Fig. 5.

A comparison of Figs. 4 and 5 shows that the habitat-related risk has become more significant even though it is still less that the toxic substances risk. The overall (total) risk in the river is now at a more acceptable level (Fig. 3), but it is clear that a point will be reached where the overall risk can no longer be reduced by simply managing for the most apparent stressor, i.e. the toxic substances in the river.

It has been recommended that uncertainty and variability be separated to provide greater accountability and transparency in a probabilistic assessment (Frey, 1993; EPA, 1997b). A two-dimensional Monte Carlo simulation with bootstrap sampling was performed in order to assess the impact of uncertainty in the stressor-response relationships on the 50th and 90th percentiles of the risk distribution. For the hypothetical case under discussion, it was assumed that one of the major problems in setting up a stressor-response relationship would be to establish where the no-effect (or more precisely, the undetectable effect) and unacceptable-effect levels would be. For the sake of illustration, assume that the location parameter (β) of the Weibull function would have the greatest uncertainty and that the uncertainty in β can be described by a normal distribution. The increase in uncertainty is reflected in an increase in the relative standard deviation (RSD, ratio of

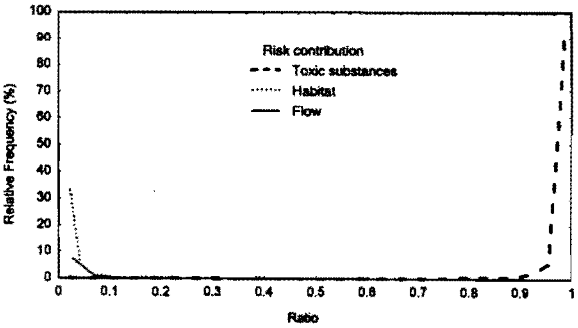

Figure 4
The ratio of stressor-specific risk to the overall risk for Scenario 1

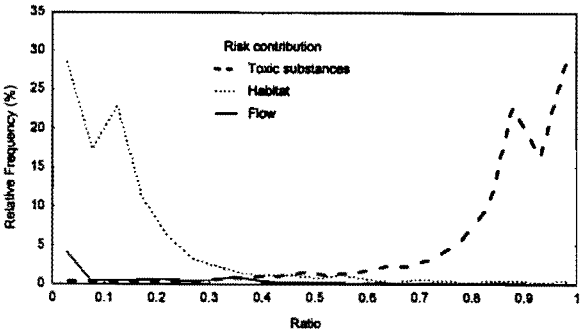

Figure 5
The ratio of stressor-specific risk to the overall risk for Scenario 4

standard deviation to median) of this uncertainty distribution. RSD values of 0.05, 0.1, 0.15 and 0.2 were used. The parameter values of Scenario 1 were used for comparative purposes. One hundred bootstrap samples from this distribution were drawn. Frey and Rhodes (1999) showed that a non-parametric method could be used in this case to select percentiles. The 50th and 95th percentiles of the overall risk distribution were established by ordering the risk values generated from 1 000 random stressor value samples and by selecting the 500th and 950th values.

From Figs. 6a and b, it is clear that there is a significant probability that the overall risk can be underestimated when there is uncertainty in the stressor-response parameters. This would, however, be dependent on the form of the stressor-response function as well as on the uncertainty distribution.

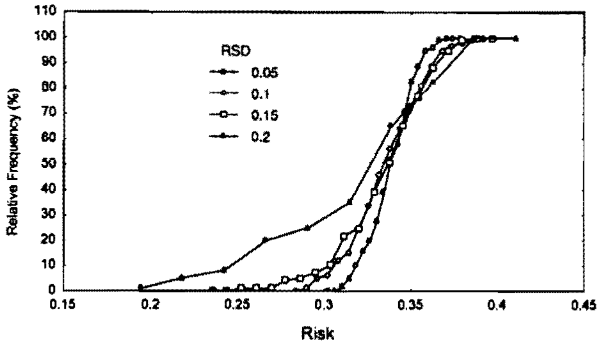

Figure 6a
The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the median risk value

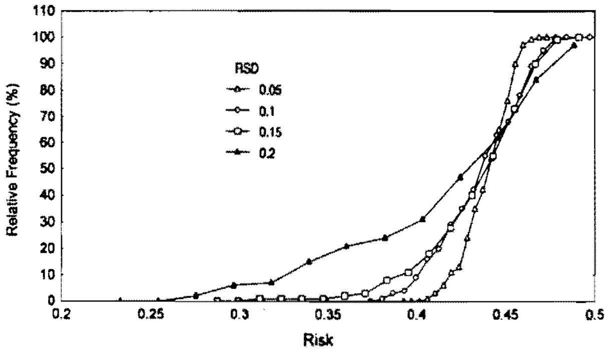

Figure 6b
The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the 95th percentile risk value.

Discussion

The left-hand side of Eq. (1) may, for example, represent the total allowable risk for a specific class of river which, in the case of the ecological reserve, may be determined by the river classification. The implication of the right-hand side of Eq. (3) is that if the individual stressor risks are defined and quantifiable, these can be managed by "trading-off" risks among stressors (as shown in the scenario exercise above) and therefore also among stressor sources. Further reduction of the risk may, for example, be effected not only by reducing the toxics concentration but also by reducing the habitat degradation. In principle, this greatly extends the management possibilities, although in practise there would likely be some bounds on the extent to which trade-offs can be accommodated, the reason being that the probabilistic approach followed here is phenomenological rather than mechanistic. Consequently, the focus is more on the expectation of an effect than on the mechanisms that caused the effect. At stressor levels representing high risk it becomes more critical that the stressor response relationships be well characterised due to the influence non-linearity may have on the expected stressor effect. At lower risk levels, it may well be possible to accommodate a trade-off among stressors. This could be particularly important when stressor discharge rates in a multiple discharge environment are being optimised to economic or technological constraints.

The evaluation of the terms in Eqs. (3) and (4) has been glossed over. In a highly standardised effect-scenario-driven ERA, such as that used in the European Union (Van Leeuwen, 1997), the estimate of stressor-probability profile, $P(x)$, may bear the greatest uncertainty. However, the stressor-response projection may have an equal, if not larger, impact on the overall uncertainty. The discipline of ecotoxicology needs to be used extensively to evaluate the response probability of toxics. Furthermore, the assumption of water depth as a stressor is far too simplistic to be of real value but it was used simply by way of illustration. It seems more likely that deviation from expected virgin run-off may be a stressor. However, much work is being done from which flow-related stress and flow-related stressor-response information can be drawn (e.g. King and Louw, 1998; Hughes and Münster, 1999) and some experimental and or observational data exist from which the possibility of effect can be inferred (e.g. Chessman et al., 1987; Quinn et al., 1992; Cooper, 1993; Roux and Thirion, 1993; Thirion, 1993). It appears that much more research is needed to assess effects at *ecosystem* level. Effect data for toxic substances exist mostly at the *individual organism* level and, to a lesser extent, at the population level, while effect data for the other stressors exist largely t the population and community level. However, more realistic risk assessment is still hampered by a lack of knowledge of conditional probability of effect at higher levels of organisation. As a simplification, it is often assumed that an impact at the lower level of organisation (where the data exist) necessarily implies an impact at the higher level of organisation. Consequently, the risk predicted at the lower level of organisation is at least as great as that predicted at the higher level of organisation since the probability of a logical consequent cannot be greater than that of the antecedent. Although this is a reasonable starting point, if all the interactions have not been accounted for and the conditional probabilities evaluated, this assumption could be seriously in error. As a result, the calculation above, and indeed any risk assessment based on such a premise, could be seriously in error.

Probability as an epistemic issue

Interpretation of the terms "risk" and "probability" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997). The interpretation of probability is crucial to decision-making in data-poor ecological management situations. The "frequentist" approach (Jaynes, 1996), sees probability as the limiting frequency of an occurrence over a large number of observations.

In contrast, probability can be seen as a subjective expression (not necessarily dependent on repetitive observations) needed to project from the domain of uncertainty by the means of prevision to the domain of certainty. "Prevision, ... consists in considering, after careful reflection, all the possible alternatives, in order to distribute among them, in the way which will appear most appropriate, one's own expectations, one's own sensations of probability" (DeFinetti, 1990). With this view in mind, probability, and by association risk, could be seen as epistemic of the specific combination of situation and assessor.

Regulatory decision-making in the field of ecology is largely dependent on a descriptive conceptual knowledge of ecosystems, often only supported by patchy observation. Observations of multiple replicates of experiments are often not available or simply impossible. What often needs to be considered is the expert prevision pertaining to a specific situation. Predictive ecological risk is essentially an expectation of an effect, a prevision based on best available knowledge of the assessor's knowledge of and expertise in dealing with, what are as yet, unobserved events in a complex system. The calculated ecological risk values are therefore an expression of the assessor's expectation, taking into consideration the scientific information at hand.

Possibility theory (based on fuzzy set theory) (DuBois and Prade, 1988) may be better suited to the kind of situation where semi-quantitative expert opinion, such as in ecology, is the basis of the decision-making process. A fuzzy mathematical approach to ecological risk has been used (e.g. Ferson and Kuhn, 1992; Ferson, 1994) and possibility theory merits investigation as a total risk estimation tool.

Conclusion

Modelling the total ecological risk as the disjunction of independent individual stressor risks can be applied to the management of a water resource subject to diverse stressors. A risk-based approach (as compared to a hazard-based approach) affords greater flexibility to the management of diverse stressor sources by maintaining a common basis for comparing the various stressors and thus creating the opportunity of prioritising and "trading" among stressor scenarios. At the same time the overall risk can be related to management classification of a water resource, providing a basis for developing class-related stressor criteria on a site-specific basis.

It is a truism that the quality of the predicted risk can be no better than that allowed by the information on which it was based. Clearly, research invested into improvement of both the ecosystem inference models and the mechanistic stressor-response and stressor-prediction models will improve the resource management flexibility.

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A possibilistic approach to diverse-stressor aquatic ecological risk estimation

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Abstract

A possibilistic approach to assess the risk of co-occurring stressors in an aquatic ecosystem based on the use of fuzzy sets is illustrated at the hand of a hypothetical case study. There are two aspects of importance: a fuzzy stressor response relationship where the response may have reference to a lower level end-point, and a rule-based inference model relating the occurrence of low-level stressors to a high-level ecological goal such as sustainability. The stressor-response is expressed as a conditional possibility. The possibility and necessity measures of the disjunctive composition of the stressor-response with the possibility distribution of the stressors yield an estimate of the ecological risk. Such a possibilistic approach may well serve as a screening procedure in multiple stressor resource management when only qualitative risk assessments are needed.

Introduction

The South African National Water Act places a premium on water supply for basic human needs and for the sustainable development and use of the aquatic ecosystem. This is reflected in the reserve. The ecological component of the reserve has been defined as that level of quantity and quality necessary to ensure the sustainable development of the water resource (NWA, 1998). The ecological reserve is a water resource management instrument for aquatic ecosystem protection to ensure sustainability in the use and development of the water resource. As a practical management measure, the capacity of the water resource to maintain its sustainability can be discretised into different management classes (MacKay, 1998) corresponding to different levels of risk that the resource may lose its sustainability.

Risk is used here in the sense of the likelihood that a specific undesired event would occur. This likelihood may be expressed in terms of either probability or possibility. In probabilistic risk assessment, it is assumed that this event is crisply defined, i.e. it is possible to decide whether the event has occurred or not. However, the nature and epistemology of the event would determine how likelihood is expressed. Possibility theory offers the option of addressing fuzzy events where the event is perhaps epistemologically vague.

A point of departure in this paper is the recognition that in assessing the risk of the aquatic ecosystem losing its sustainability:

- there are several stressors (such as chemical substances, flow reduction and habitat degradation) that may be present simultaneously and that may result in responses such as loss of sustainability (although the mechanics of these impacts may differ), and
- unambiguous quantitative and possibly even quantitative site specific data may often be lacking.

An argument will be presented for the application of a fuzzy approach to aquatic ecological risk. Two types of ecological risk may be defined depending on how the likelihood measure is expressed: a risk based on a possibility measure (referred to as "ecological concern") and a risk based on a necessity measure (related to the possibility measure and referred to as "ecological dread"). These are illustrated by a hypothetical application to water resource classification.

Rationale for a fuzzy approach

The term "sustainability" is not defined in the NWA. For the purpose of discussion, it is assumed that ecological sustainability refers to the ability of a system to maintain an acceptable level integrity subject to anthropogenic stress. Concepts such as sustainability and integrity may be spatially and temporally scale-dependent and the knowledge of the mechanisms underpinning these phenomena is vague (Costanza et al., 1993, De Leo and Levin, 1997). Variability is both a normal and sometimes a necessary ecosystem characteristic to certain ecosystem processes. "Therefore, in managing ecosystems, the goal should not be to eliminate all forms of disturbance, but rather to maintain processes within limits or ranges of variation that may be considered natural, historic or acceptable" (De Leo and Levin, 1997).

Not only must natural variability be accounted for in the management process, but also uncertainty and, in some cases, vagueness. Definitions of ecosystem integrity varies; e.g. "the maintenance of the community structure and function characteristic of a particular locale or deemed satisfactory to society" (Cairns, 1977) or "the capability of supporting and maintaining a balanced, integrative, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley, 1981). Terms such as "deemed satisfactory"; "balanced", "comparable" and "natural" in these definitions are, without further qualification, essentially vague and subjective. This means that in terms of the risk assessment under the NWA, the end-point is vague.

In addition, the system boundaries, the response to stressors and the stressors themselves may only be known qualitatively. The functional entities that best reflect the goals of ecosystem management may only be vaguely identifiable. Consequently, in dealing with ecological risk in the context of protective ecosystem management, it would be advantageous to use a paradigm that is

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adapted to address both uncertainty and vagueness. This could be accomplished by using the framework of possibility theory (as opposed to probability theory), which is based on the use of fuzzy logic (as opposed to 'crisp' logic).

Probabilistic vs. fuzzy risk

Risk is a way of expressing the uncertainty of observing some event (Suter, 1993). The use of risk techniques in decision-making is largely motivated by the variability and uncertainty observed in dealing with ecosystems and has been used extensively in a number of countries (e.g. USEPA, 1996; Pederson, et al., 1995). Probabilistic risk assessment depends crucially on the ability to derive some expression of probability for a stressor variable. Conventionally, imperfect information has been dealt with either by probability or by interval analysis.

Probability theory has, over a period of 200 years, developed a calculus to deal with stochasticity. A problem with probability theory in ecological risk assessment may relate to the interpretation of what is really represented by probability (Dubois and Prade, 1988). The frequentist approach sees probability as the *limiting frequency* of observed, *clearly defined* events. The first major obstacle in assigning probability distributions for ecological variables is the lack of enough system-specific information to estimate these limiting frequencies. The alternative Bayesian approach circumvents the frequentist dilemma by using probability as a descriptor of the state of knowledge about an event or proposition (Jaynes, 1996) and is often much better suited to generating the necessary distribution data.

The second (and possibly more critical) problem facing ecological risk assessment and risk management is the difficulty in defining the system uniquely at an operational level. The boundaries of ecosystems, communities and even populations, for example, are notoriously vague. This complicates the use of both frequentist and Bayesian statistics, which deal with such vagueness with difficulty. Mathematically, this vagueness, superimposed on the complexity of ecosystems, the elements of which may exhibit stochastic behaviour, results in analyses that become intractable to conventional mathematics. The resulting ecosystem models exist largely as lexical system descriptions. In analyzing a complex multidimensional system, a state could be reached where, even if uncertainty and variability could be quantified, the results would be difficult to interpret (Dubois and Prade, 1988). As the complexity of the system or model of a system increases, a point could be reached where "our ability to make precise and yet significant statements about its behaviour diminishes until a threshold is reached beyond which precision (or relevance) becomes almost mutually exclusive characteristics" (Zadeh, 1973)

Working with incomplete data, ecologists may have to deal largely with judgement, which by its nature has at least an element (if not consisting entirely) of subjective opinion. Possibility theory in contrast to probability theory, "offers a model for the quantification of judgement which allows a canonical generalisation of interval analysis" (Dubois and Prade, 1988) which has been used in the analysis of uncertainty in the physical sciences.

Risk estimation in ecosystems has been shown to be influenced by both uncertainty and variability (e.g. Frey, 1993, Frey and Rhodes, 1999), which argues for a probabilistic rather than a deterministic approach in assessment. The concept of risk contains the elements of:

- value ("what is being threatened"),
- extent ("how badly"),
- the likelihood of a) and b), and
- assessment ("what does it mean").

Applying possibility theory to assessment of ecological reserve-related risk

For discrete events ω with a possibility distribution $\pi(\omega)$, the possibility measure $Poss(A)$ and the necessity measure $Nec(A)$ are defined by Eq.1.

$$\begin{aligned} Poss(A) &= \sup\{\pi(\omega) \mid \omega \in A\} \\ Nec(A) &= \inf\{1-\pi(\omega) \mid \omega \notin A\} \end{aligned} \tag{1}$$

Some of the differences between probability measures and possibility or necessity measures are:

- The *probability* of the sure event is assigned the value 1. For a number of events, the cumulative probability of all possible events is assigned the value 1. A *possibility* of 1, however, does not imply that the event is sure, only that it is entirely possible.
- The knowledge of the probability of an event completely determines the knowledge of the contrary event. Knowledge of the possibility or necessity of an event is less strongly linked to the knowledge of the contrary event. To establish the certainty of an event, it is necessary to know both the possibility and the necessity of the event.
- Probability deals with precise but differentiated items of information. Possibility reflects imprecise but coherent items.
- A central requirement in probability theory is the additivity of the probability of independent, mutually exclusive (disjoint) events. This requirement, generally, does not hold for fuzzy likelihood measures.

These characteristics of possibility theory make possibility measures well-suited to reasoning in an uncertain environment where it is often desirable not to set the relationship between the evidence one has for an event (degree of necessity) and the evidence that weighs against it (1-degree of possibility) too rigidly. In addition, it might be prudent to consider whether one's knowledge that an event (such as loss of sustainability) might occur, also defines the possibility that the event might not occur. In other words, does one's *knowledge of the ecosystem* allow for the law of the excluded middle of Aristotelian ('crisp') logic?

Variability: an inherent and practically irreducible characteristic of a biotic system, stemming from the innate stochasticity underlying processes in the ecosystem.
Uncertainty: epistemic of the observer stemming from imperfect information, due to limitations in observation, modelling or interpretation of system-related data, for example.
Vagueness (or fuzziness): a lack of clarity in the definition of the set of values attached to the object.
Ambiguity: largely associated with language, where the definition of the object is vague or refers to several different reference sets simultaneously.

The regulatory end-point E

In an ecological risk assessment implicit in the classification in terms of the reserve, the "regulatory" undesired event, E, is defined by the NWA as "loss of sustainability". This is a fuzzy event in the light of the foregoing. The management classes in the NWA correspond to differences in the likelihood of this fuzzy event occurring.

This definition of E implies that it is a dichotomous characteristic of the system; anything less than full sustainability means unsustainability. It does not mean that important related characteristics such as resilience and integrity need to be dichotomous as well. There might be levels of resilience and integrity less than 100% that still result in sustainability. E may be epistemologically vague, in that the knowledge of what constitutes E (or $\neg E$ i.e. "not E") may be imperfect. An assessment of the "likelihood of E" may be a reflection of the epistemology of the values of the parameters defining the critical point defining E. Consequently, the evidence one has that a certain set of parameter values corresponds to E and the evidence that it corresponds to $\neg E$ might not be complementary in the sense that one's knowledge of E occurring does not define one's knowledge of E not occurring. There might, therefore, be a set of parameter values for which it is not possible to make a clear assessment of either the likelihood of E or the likelihood of $\neg E$. The "likelihood of E" is interpreted as the degree to which the observed situation corresponds to E.

Ecological concern and dread

The likelihood aspect of risk can be expressed in terms of possibility theoretical concepts. *Poss* (E) could be used to express the possibility that effect E would occur. This does not carry the same weight as the probability of E, P(E). It is always true that $Nec(E) \leq P(E) \leq Poss(E)$. This means that Poss ($E$) expresses an epistemic possibility that E could occur and therefore, *Poss* (E) expresses a weaker claim than P(E). More appropriately, *Poss* (E) might designate the degree of "ecological concern".

On the other hand, *Nec*(E) expresses the cumulative evidence of the necessity that E must occur. This is a much stronger claim that P(E) and may appropriately be expressed as the degree of "ecological dread". Both ecological concern and ecological dread express the accumulated evidence about the likelihood that the undesired event E will occur.

There are three aspects to the assessment of ecological risk in the aquatic environment that are important in the context of the reserve:

- The estimation of the aggregate likelihood of *Poss*(E) or *Nec*(E) when diverse stressors occur together,
- The confidence in *Poss* (E) or *Nec*(E) on projecting E from other available data and
- The formulation of the relationship between *Poss* (E) or *Nec*(E) and the stressor value.

Aggregating diverse stressors

There are a number of different stressors that could result in loss of sustainability. Assume, for example, that flow deficiency (i.e. degree to which the flow is less than that expected in the natural hydrological cycle), toxic substances and habitat degradation are typical stressors in a system being assessed. In order for E to occur, it is assumed that:

- An environmental variable X with value x, only becomes a stressor if it can result in E, i.e. in the present context, stress is not defined if a variable is within its natural range of variability. Furthermore, there exists a critical value x_0 at which E occurs. Our knowledge (rather than the inherent nature) of E as well as x_0 make both fuzzy quantities. The likelihood of E occurring (both *Poss* (E) or *Nec*(E)) is a function of x. The stress E_X, which is used here in the sense of the extent of the effect E being produced as a result of stressor X, depends on a fuzzy causal relationship $E|X$ and an occurrence of stressor X, where the X is a fuzzy set of stressor values which correspond to x_0 and which is defined in terms of the degree to which a value x corresponds to $x_0 : X = \{x|\ \mu_X(x) = \pi(x{=}x_0)\}$.
- Any of the stressors could result in E, irrespective of whether they occur alone or together. The *ecological concern* would refer to the *possibility* that *any* of the stressors (and by implication the resultant stresses) occur. The *ecological dread* would refer to the *necessity* that *all* the stressors occur together (in which case there is no doubt in the assessor's mind that E is likely to occur).
- Generally, it would not be known (at least at the outset) whether there is an additive, supra-additive ("synergistic") or infra-additive ("antagonistic") interaction among stressors. The way in which this is approached is largely a matter of assumption until further evidence is produced. The assumption will be reflected in the risk aggregation operators (t-norm and t-conorm in Eq. (3) below).

For the stressors noted above, these assumptions could be interpreted as:

- There exists a value of flow, q_0, in a given river section, for example, which will result in loss of sustainability if this flow is maintained for a sufficient period. Although the exact value is unknown, flow requirement studies (e.g. King and Louw, 1998) may yield some idea of what it might be. The flow-related concern and dread for any specific value of flow, q, under discussion, can be estimated from Eq. (2a):

$$\begin{aligned} Poss(E_Q) &= Poss((E\mid Q)\wedge Q) = t-norm\{Poss(E\mid Q), Poss(Q)\} \\ Nec(E_Q) &= t-norm\{Nec(E\mid Q), Nec(Q)\} \end{aligned} \tag{2a}$$

- There exists a critical value of toxic substance concentration, t_0, (as toxicity units) such that for any specific value t the toxics-related concern and dread would be given by Eq. (2b).

$$\begin{aligned} Poss(E_T) &= Poss((E\mid T)\wedge T) = t-norm\{Poss(E\mid T), poss(T)\} \\ Nec(E_T) &= t-norm\{Nec(E\mid T), Nec(T)\} \end{aligned} \tag{2b}$$

- Analogous to the above, the fuzzy critical habitat degradation value H is assessed by expert opinion so that for any specific level of habitat degradation, h, the habitat-related concern and dread will be given by Eq. (2c).

$$\begin{aligned} Poss(E_H) &= Poss((E\mid h_0)\wedge H) = t-norm\{Poss(E\mid H), Poss(H)\} \\ Nec(E_H) &= t-norm\{Nec(E\mid H), Nec(H)\} \end{aligned} \tag{2c}$$

The fuzzy set X is normalised since by assumption a stressor is only defined as such if there is at least one value of x such that $\mu_X(x) = 1$, i.e. there is at least one value for which E is entirely possible. Hence, the equivalence of the membership function values with the possibility of X.

A further result of the assumptions above is that the ecological concern ρ_c and ecological dread ρ_d is expressed in Eq. (3):

TABLE 1
Some possible *t-norms* and *–conorms* (Kruse et al., 1994) for use as aggregation operators on quantities *a* and *b* in assessing ρ_c and ρ_d

Type	t-norm	t-conorm	Implication ($\alpha \rightarrow \beta$)	Interpretation
Min-max(a,b)	Min$\{a,b\}$	Max$\{a,b\}$	Min$\{1-\alpha+\beta, 1\}$	Components contribute independently
Lukasiewicz(a,b)	Max$\{0, a+b-1\}$	Min$\{a+b, 1\}$	$\begin{cases} 1 \ \mathit{if}\ \alpha \le \beta \\ \beta \ \mathit{otherwise} \end{cases}$	Components additive
Probabilistic(a.b)	$a.b$	$a+b-ab$	$\begin{cases} \frac{\beta}{\alpha} \ \mathit{if}\ \beta < \alpha \\ 1 \ \mathit{otherwise} \end{cases}$	Intermediate between min-max and Lukasiewicz

$$\begin{aligned} \rho_c = Poss(E) &= Poss(E_Q \vee E_T \vee E_H) = t-conorm\{Poss(E_Q), Poss(E_T), Poss(E_H)\} \\ &= \min\left\{\sum_{X \in \{Q,T,H\}} Poss(E_X), 1\right\} \\ \rho_d = Nec(E) &= Nec(E_Q \wedge E_T \wedge E_H) = t-norm\{Nec(E_Q), Nec(E_T), Nec(E_H)\} \\ &= \max\left\{0, \sum_{X \in \{Q,T,H\}} Nec(E_X)\right\} \end{aligned} \tag{3}$$

The implication is that if $\rho_c = 0$ then E is considered impossible (inasmuch as our knowledge base allows for that) and $\rho_d = 0$ by definition. If $\rho_c = 1$, then E is considered entirely possible (of course not necessarily entirely probable) and ρ_d may be ≥ 0, which means that not only is E possible, but it may also necessarily occur. If $0 < \rho_c < 1$, then E is possible to the extent ρ_c but $\rho_d = 0$ (if an event is not entirely possible it cannot be at all necessary).

The choice of *t-norm* and *t-conorm* in Eq. (3) for the stress aggregation needs to take cognisance of the knowledge about the interaction among stresses. For toxic substances, true synergism among the substances appears to be rare (Hermens et al., 1984a; 1984b; Calamari and Vighi, 1992) although it has been reported (Broderius and Kahl, 1985). Additivity of toxicity occurs more often than true synergism or supra-additivity. For other stressors, effects such as additivity have not been reported on if they do exist. Even less so has synergism among diverse stressors been reported on.

It is likely that additivity of effect among diverse stressors reflects the worst case, while additivity may also be possible. Some of the possible *t-norms* and *–conorms* that could be used in aggregating fuzzy risks are listed in Table 1.

For the aggregation of concern and dread (Eq. (3)) the Lukasiewicz aggregation with the implied additivity of stresses appears to the most conservative option. For the aggregation of risk components (Eqs. (2a) to (2c)), exposure and effect may be seen as contributing independently to the likelihood of effect, and consequently, the min-max aggregation would be more suitable.

End-point projection

The regulatory end-point E, which is at ecosystem scale, is unlikely to have data at the correct spatial and temporal scale from which it can be derived. It is more likely that, on a case-specific basis, phenomena at smaller spatial and temporal scales will be used to infer the occurrence of E. Lower level phenomena such as the disappearance of key species, loss of integrity, mortality of selected species are more likely to be used to infer E.

For example, assume that a toxic substance is introduced into a river system. From toxicity assessment it might be established that if the concentration of the toxic substance is x then the cumulative probability of an individual in a population of the test species Z will die, is y, with confidence interval (y_1, y_2). The toxicity concern, $Poss(E_T)$, and dread, $Nec(E_T)$, must be estimated from these data. In order to do this, it is necessary to follow some conceptual inference model such as Eq. (4)

Rule base (R):

IF concentration IS x THEN an individual of species Z IS dead (Possibility = y_1)
IF an individual of species Z IS dead THEN the population of Z IS lost (Possibility = α)
IF the population of Z IS lost THEN a key species IS lost (Possibility = β) (4)
IF a key species IS lost THEN integrity of the ecosystem IS irreversibly compromised (Possibility = γ)
IF integrity of the ecosystem IS irreversibly compromised THEN sustainability IS lost (Possibility = δ)
Observation (X): The concentration IS x (Possibility = ε)

An analogous rule base can be formulated for N(E_T). The value of *Poss* (E_T) derives from the conjunction $R \wedge P$. This value will be a function of y_1, α, β, γ, δ and ε. In its simplest form *Poss* (E_T) $\leq \min\{y_1, \alpha, \beta, \gamma, \delta, \varepsilon\}$. (For $Nec(E_T)$ the inequality will be replaced by an equality.) This would support the notion that the possibility that toxics cause a loss of sustainability can be no stronger than the weakest inferential link. Since specific data for their assessment is usually lacking, the values for α, β, γ, δ and ε may conservatively be set equal to 1. The assumption should not simply be made that the confidence in the lower level phenomenon is equal to that of E (Suter, 1993; 1995).

Stressor-response relationships

A crucial component of the individual stressor concern (or dread) assessment is the conditional term *Poss* ($E|x_0$) or $Nec(E|x_0)$. These terms are essentially the output of the effect assessment phase of an ecological risk assessment in the context where an end-point is fixed. It summarises the knowledge about the expectation of effect of the stressor on the system being assessed and answers the implied question: "What if the system is being exposed to the stressor"? In the present context, both E and x_0 are fuzzy entities and, hence, the condition term represents a fuzzy relationship, R_X. R_X is the formalised knowledge base on the relationship between the likelihood of E and x. The likelihood of individual stresses is derived from R_X and an observation X by Eq. (5). An expression for $Nec(E_X)$ can be similarly derived from Eq. (1).

$$\begin{aligned} Poss(E_X) &= R_X \overset{t}{\circ} X = \sup_x\{t-norm\{\mu_X(x), R_X(E,x)\}\} \\ &= \sup_x\{\min\{\mu_X(x), R_X(E,x)\}\} \end{aligned} \tag{5}$$

The relationship R_X derives from a rule-base of the kind "If $X=x$ then $E = \varepsilon$" where the truth-value of $X=x$ is $\mu_X(x)$ and that of $E = \varepsilon$ is $\mu_E(\varepsilon)$. This can then be formulated as "$\mu_X(x) \rightarrow \mu'_E(\varepsilon)$". Using the max-min implication (Table 1) Eq. (5) becomes Eq. (6).

$$Poss(E_X) = \sup_x \left\{ \min \left\{ \mu_X(x), \min_\varepsilon \{1 - \mu_X(x) + \mu_E(\varepsilon), 1\} \right\} \right\} \quad (6)$$

Evaluating R_X now becomes the problem of evaluating the relationship $\mu_X(x) \rightarrow \mu_E(\varepsilon)$, or "IF $\mu_X(x)$ THEN $\mu_E(\varepsilon)$". There are two distinct ways to generate this assessment:

- Cause to effect: Given a stressor value x, to what extent will its impact comply to the description E (i.e. $x \rightarrow E$) and
- Effect to cause: Given a level of effect ε, what are the levels of x that correspond to ε (i.e. $E \rightarrow x$).

In general, this need not be a mathematical-functional relationship. If the best knowledge available is in the form of fuzzy "rules" such as those in Eq. (4), then the stressor-response relationship (SRR) is at best a fuzzy mapping of the stressor value domain to the response likelihood domain.

Hypothetical case study

In an ecological risk assessment study, it is agreed that there are three major stressors in a catchment, i.e., unidentified toxic substances, deviation from expected flow and physical habitat degradation. There are three types of information that is required from expert input:

- Definition of the SRR from a) the lowest stressor value where effect E may be expected to be discernable (x_{11}), b) the lowest stressor value where E may be entirely possible (x_{12}), c) the highest level where E may be discernable (x_{21}) and d) the highest level where E is entirely possible (x_{22}).
- The epistemological confidence on projecting from the observable response to the regulatory end-point (α, β, γ, δ and ε).
- The likelihood of the occurrence of the stressor. ($\mu_X(x)$)

Fuzzification of concern and dread

Consider a situation in a river system where the critical effect, E, being assessed is "loss of sustainability". Due to the epistemic uncertainty relating to mechanisms, thresholds, subjectivity in assessments, etc. in a river system, the risk of E (expressed here as the possibility of E) is described in terms of categories rather than numerical terms. For example, the level of risk may be assessed as belonging to a class K such that the set K = {Insignificant, Low, Marginal, Significant and High} as shown in Fig. 1.

These classes are vague since their boundaries may be a matter of interpretation. An effect possibility of 0.2 might be described as being 'low' or 'marginal' to some extent. Consequently, the classes are modelled as fuzzy sets. These same 'fuzzification' parameters might also be used in describing the concern and dread levels since they deal with the same type of possibilistic measures.

The definition of individual stressor effect possibility (Eq. (6)), as well as the aggregated concern values (Eq. (3)), ensure that at least one of the fuzzy sets will have a membership value of 1. This means that it will be possible to describe the concern level in a river or stream in terms of at least one of the classes. However, it may be possible that more than one class has a membership of 1, in which case the worst class that has a membership of 1 will logically be class descriptor for the river situation.

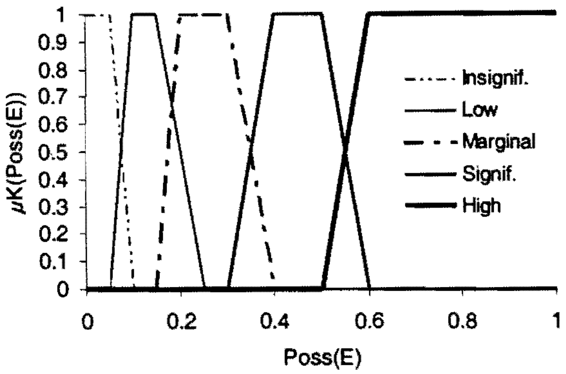

Figure 1
The parameters for describing the possibility of E *in terms of the set* K *of fuzzy labels. The fuzzy set is defined by the degree to which the possibility of effect,* Poss*(E), corresponds to the descriptor* K.

Toxic stress

The toxicity stress is determined by toxicity bioassessment studies without specifically identifying the toxic components and is expressed as toxicity units. A toxicity unit is defined as 100 divided by a benchmark effect concentration expressed as a percentage of the effluent. The data are derived from single species toxicity tests and projections of effect to population level (e.g. Caswell, 1989). The no-observable-effect concentration (NOEC) is taken to be at 10% of the EC50.

In-stream objectives of 0.3 TUa and 1 TUa have been suggested as levels where no critical effects should be observed (USEPA, 1991, Tonkes and Balthus, 1997) and these values are used for x_{11} and x_{21} respectively. It is assumed that at double the EC50, sustainability might be lost if predation pressure is high while, even under the best circumstances, sustainability is in jeopardy if 99% (corresponding perhaps to 3 times the EC50) of a population dies. These values are used for x_{12} and x_{22} respectively (Fig. 2(a)). The possibility distribution for x is assumed to be a triangular distribution such that $\mu_X(x) = 0$ corresponds to the 5th and 95th percentile values while $\mu_X(x) = 1$ corresponds to the median value. The values of α, β, γ, δ and ε in Eq. (4) are all assumed to be 1.

Flow stress

The flow stress, q, is assumed to be due to the reduction of the expected flow in stream. The value of $q = 0$ when the stream flow is very similar to pristine flow while $q = 1$ corresponds to critical disruption of stream flow. The values for the mapping parameters are entirely hypothetical (Fig. 2(b)).

Habitat stress

The habitat degradation is assessed by a river ecologist and expressed as a percentage deviation from what is expected to be pristine. The values for the mapping parameters are entirely hypothetical (Fig. 2(c)). The fuzzy relationships were assumed to show a triangular distribution such that for any stressor level, the effect is given by a triangular distribution with its least likely values given by y_1 and y_2 (see Appendix) and its most likely value by y_m.

TABLE 2
The scenarios in which the ecological risk assessment is evaluated.

Scenario	Toxic substance status	Flow status	Habitat status
1	The levels are practically pristine. Discharges are mostly assimilated	Very little abstraction or water loss is evident. Sporadic abstraction has a minor impact.	Practically pristine. Only minor modifications (10%) are found.
2	Substantial discharges exist. With a very low frequency up to 5 TUa is found while there is usually some chronic toxicity detected (0.1 TUa). Values of 1 TUa is found commonly.	Extensive abstraction takes place at times resulting stressor levels within 20% of critical levels. On rare occasions the flow is practically pristine, but mostly the flow is within 50% of pristine.	There is almost no pristine habitat left with some areas being largely modified (about 75%) while most of the stream has about 50% suitable habitat left.
3	Rigid control on point sources is instituted but on rare occasions 1 TUa is still found, but mostly toxicity is around 0.3TUa or even as low as no detectable toxicity.	Some control on abstraction is possible and flows within 20% of expected can often be achieved. However, on rare occasions up to 80% of the pristine flow is abstracted.	Some habitat remediation could be effected so that most of the river now has 25% loss of the pristine habitat while the worst case has only about 50%.
4	Toxicity is managed to be around 0.55TUa most of the time while excursions up to 1.1 are rarely found.	Same as in 1.	Same as in 1.

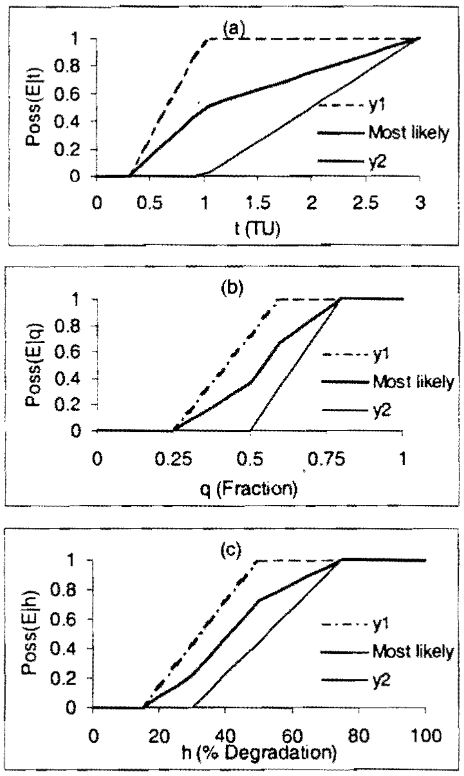

Figure 2
The fuzzy mapping representing the SRR's for the stressors in this study: (a) SRR for toxicity stress, (b) SRR for flow stress and (c) SRR for habitat stress.

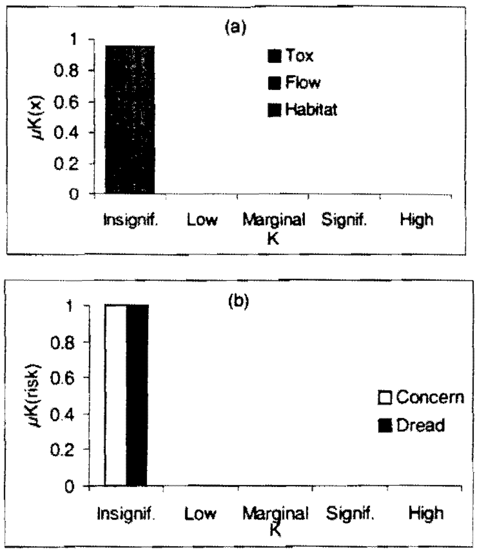

Figure 3
The classification of (a) the stressor specific possibility of effect in terms of fuzzy set membership to the class K (see Fig. 1) and (b) the concern and dread for Scenario 1 (Table 2).

Methodology

Eqs. (1) to (3), (5) and (6) as well as those in the Appendix were solved using an Excel97 spreadsheet under Windows 95.

The use of ecological concern and ecological dread was investigated by considering its value in four scenarios as described in Table 2.

The narrative description of scenario 1 in Table 2 yields

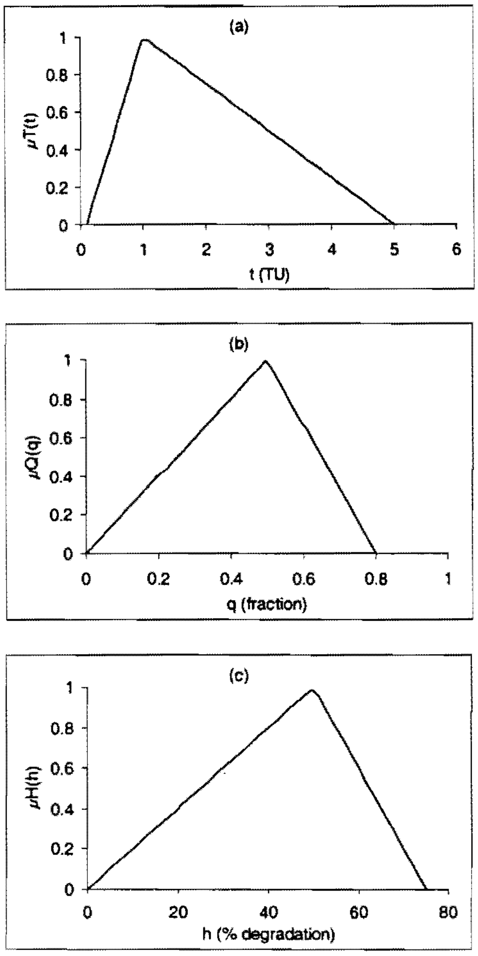

Figure 4
The stressor possibility distributions for (a) toxics-, (b) flow- and (c) habitat-related stress (expressed as $\mu_x(x)$) derived from the descriptive data for Scenario 2 in Table 2

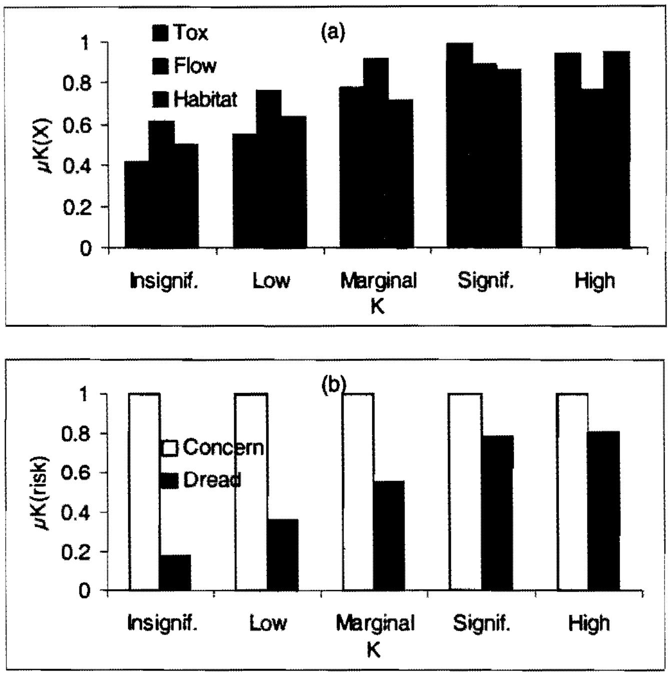

Figure 5
The classification in terms of class K (Fig. 1) of (a) stressor specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 2

stressor possibility distributions that are triangular with vertex at (0,0). The stressor possibility distributions for scenario's 2 and 3 are shown in Figs. 4 and 6 while the SRR's are shown in Fig. 2.

Results and discussion

The individual stressor risks are shown in Figs. 3, 5, 7 and 8.

Scenarios 1 to 3 were chosen to represent a pristine, a heavily utilised and a reasonably managed system respectively. The pristine system, not surprisingly, yielded an assessment of insignificant risk for each individual stressor (Fig. 3(a)). Consequently, both the concern and dread (Fig. 3(b)) are 'insignificant' as would be expected.

In the case of the heavily utilised system (Fig. 5) the individual stressor risk values are either 'significant' (toxics and flow) or 'high' (habitat), considering the maximum membership values. The aggregation method used here results in a concern membership value of 1 to all classes. Since the worst class will reasonably dominate, it could be said that the concern level is 'high'. In this case the dread value is used to distinguish between the classes, so that a dread class of 'high' could be allocated.

An analysis of the stressor risk contributions in Scenario 2 shows that all the stressors need attenuation. It is assumed that in the managed system (Scenario 3) it is possible to manage the discharge of toxics as well as abstractions to a reasonable extent while stream habitat remediation is less successful (Fig. 6). The results (Fig. 7) indicate that although toxic and flow risk are now largely 'insignificant' and habitat risk is 'low', on aggregate the concern level is still no better than 'high'. The dread value though has become ''insignificant', demonstrating that progress had been made in improving the situation.

Scenario 4 (Fig. 8) was used to illustrate a possible use of concern and dread assessment in assessing the change in criteria (in this case the example of toxicity management criteria). It was now assumed that both habitat and flow risk were insignificant. By systematically changing of the most likely value and the upper limit value in the toxicity possibility distribution, it was attempted to find a parameter set that would be on the verge of changing the concern assessment from 'insignificant' to 'low'. This parameter set is reflected in Table 2. This is in spite of the toxic effect possibility being 'low' or even 'marginal'.

The interpretation of Scenario 4 is that if there are no other stressors that could significantly contribute to the ecological risk, then the parameter values for this scenario will be the maximum allowable to maintain 'insignificant' concern and dread levels.

It has been assumed that risk objectives for the river have been set. This is generally not true for South African rivers. The parameters (i.e. the *Poss*(E) values defining the fuzzy set trapezium in Fig.1) used for classifying response possibility are critical. In this hypothetical study the fuzzification as depicted in Fig. 1was simply assumed. No formal procedure was put forward to derive rational values for these parameters and this aspect needs more

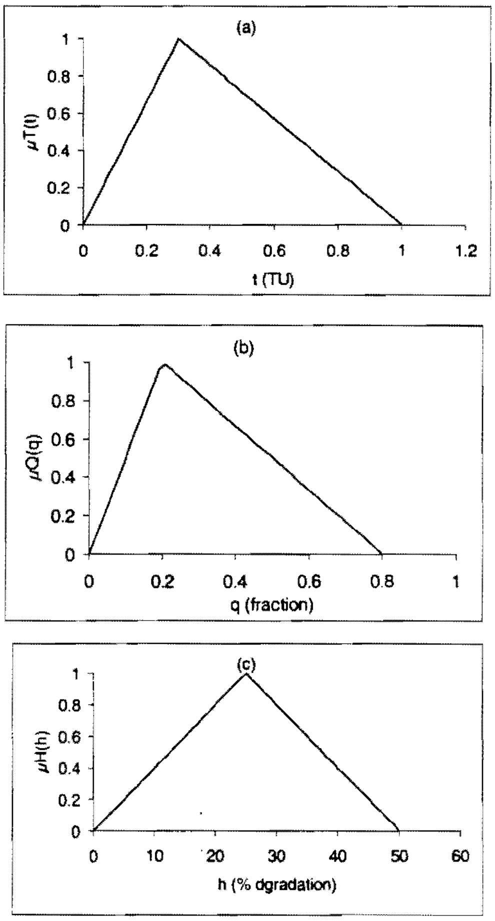

Figure 6
The stressor possibility distributions for (a) toxics-, (b) flow- and (c) habitat-related stress (expressed as $\mu_x(x)$) derived from the descriptive data for Scenario 3 in Table 2

extensive consideration. Any procedure for deriving the fuzzification parameters would have to take cognisance of:

- correspondence between observed system assessments and the concern and risk classes projections, and
- the risk perceptions of the user community.

The former problem can probably best be addressed by analysis of a database containing both bio-monitoring and stressor data by a tool such as neural networks. The assumption is that the concern and dread levels will generally be reflected in the trends in stream bio-integrity. The latter problem is similar to the domain of risk communication except that risk values are usually in probabilistic rather than possibilistic terms.

The concern and dread assessments are also significantly affected by the SRRs. The use of fuzzy mapping as SRRs addresses this problem to some extent. With reference to SRRs it is noted that:

- If the uncertainty in the different SRRs differ widely, it is apparent that the higher uncertainty will dominate the assessment uncertainty. It may, for example, be unnecessary to insist on high confidence toxicity response data (simply because it can

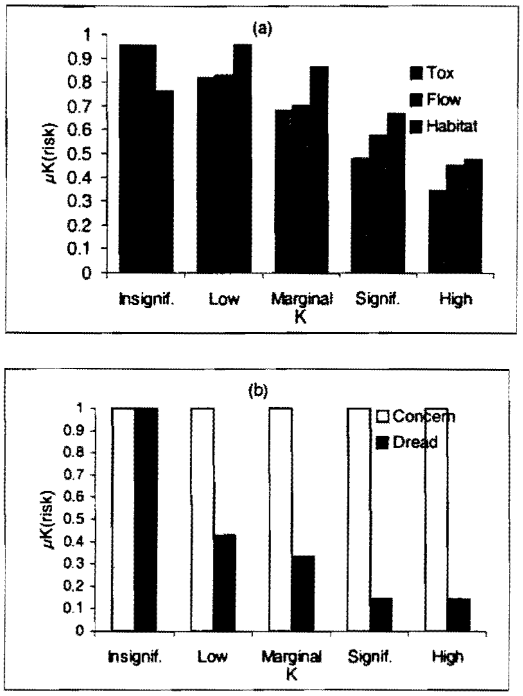

Figure 7
The classification in terms of class K (Fig. 1) of (a) stressor-specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 3

be achieved), while having to accept very coarse data on habitat related stressor-response information.

- It has tacitly been assumed that the identification of a stressor had taken into consideration a temporal component if at all applicable. It is known that toxic substances may accumulate over a period to toxic levels in an organism (e.g. Mancini, 1983; Legierse, et al., 1999). However, for toxic substances intra-organismal stressor exposure was assumed to be proportional to the stressor magnitude, while the temporal characteristics of the stressor had been neglected.
- In the case of flow stress, the assumption that stress is simply proportional to reduction from expected flow, is probably too simplistic. It is known that a certain amount of flow variability is both normal and necessary for the functioning of most South African aquatic ecosystems (King and Louw, 1998). A more realistic description of flow-related stress would likely involve a stochastic variable whereby the range becomes abnormal.
- The duration of stress has not been explicitly addressed for any of the stressors. This paper does not particularly concern itself with the detail of such a description, except to postulate that such a descriptor will have a magnitude component and a temporal duration component, both of which could be variable. It is possible that the variables used to characterise the stress descriptor would be crisp, but the advantage of the fuzzy approach is that they could be vague or fuzzy (depending on the state of knowledge) without invalidating the approach.

Considering Eqs.(2), (3) and (5) or (6), it is trivial to recognise that there are theoretically an infinite number of stressor-specific fuzzy risk combinations that result in the same concern (or dread) value. If only a single stressor was being addressed, it would simply

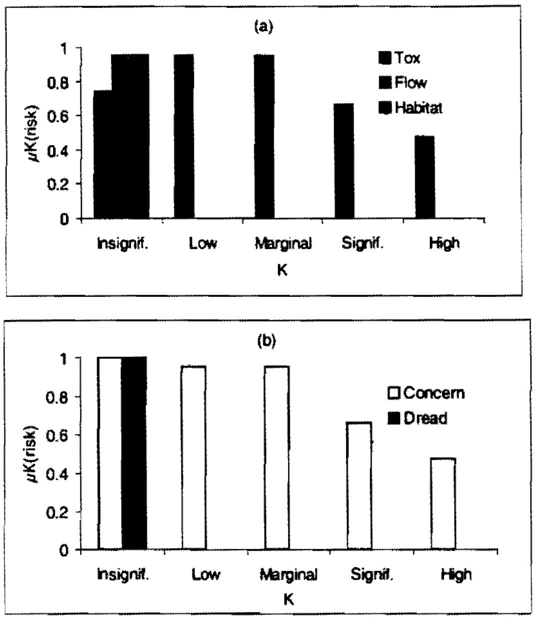

Figure 8
The classification in terms of class K (Fig. 1) of (a) stressor-specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 4

require a waste load allocation-type of calculation to distribute the stressor load among stressor sources (USEPA, 1991). However, the essence of the concern and dread calculation is the aggregtion of the diverse stressors into one measure. This means that in order to select among the infinite number of source-specific stressor-level combinations, some form of optimisation procedure would be called for. While this is a more complex task than a waste load allocation (USEPA, 1991) it also increases the management flexibility by opening the way for cost-risk-benefit calculation. This aspect requires some investigative work, although there is a substantial volume of work in the field of fuzzy optimisation (Dubois and Prade, 1994, Klir and Yuan, 1995, Sasikumar and Mujumdar, 1997).

The mathematical structure of the model is unaffected by the number of premises and propositions since it is based mostly on *max* and *min* operations. The extension to additional interactions is trivial. However, the possibility and necessity measures for the rules need to be stated as they determine the confidence in the overall assessment and this holds true for the stressor-effect implications.

Conclusions

This paper is an attempt to motivate the use of a possibilistic approach to ecological risk assessment rather than the more common probablistic approach in cases where there is epistemic uncertainty as well as stochasticity in the system being assessed. The use of fuzzy logic and a possibilistic approach to ecological risk makes use of three types of information:

- an assessment of the relationship between stressors magnitude and the expected response at a suitable level of organisation in the form of a fuzzy implication relationship,
- a possibility distribution for each stressor, and
- a logical inference model connecting direct stressor effects and the higher level end-points for the assessment in the form of a rule base.

The possibilitic ERA formulation has the advantage that it could make use of the vague information that is often all that is available for ecosystems effects, but it can also be used where precise information is available. For an application where there is no need for more precise or numeric risk data, this fuzzy set approach may be sufficient. However, the use of fuzzy variables cannot be used as a cover for bad or misleading data. The scientific quality of data is a separate issue from fuzziness. While high quality data can be fuzzified, doubtful, vague or conflicting data cannot be improved by this technique. It is necessary to be explicit with the uncertainty and vagueness in the formulation of the ecological risk assessment problem.

The parameters used in the fuzzification of data need to be considered carefully. These must be agreeable to both the risk assessor and the risk manager. This is particularly crucial where stressor response curves are very steep, i.e. where large changes in response (or fuzzy set) correspond to relatively small changes in stressor exposure.

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Appendix

The stressor response relationship for each stressor is delineated by a fuzzy mapping (See Figs. 2 to 4) such that:

$$y_1 = \begin{cases} 0 & if\ x \le x_{11} \\ \dfrac{x - x_{11}}{x_{12} - x_{11}} & if\ x_{11} < x < x_{12} \\ 1 & if\ x \ge x_{12} \end{cases}, \quad y_2 = \begin{cases} 0 & if\ x \le x_{21} \\ \dfrac{x - x_{21}}{x_{22} - x_{21}} & if\ x_{21} < x < x_{22} \\ 1 & if\ x \ge x_{22} \end{cases} \quad \text{and} \quad y_m = \frac{y_1 + y_2}{2}$$

so that $\forall y \in [y_2, y_1] Poss(E) = \begin{cases} 0 & if\ y \notin [y_1, y_2] \\ \dfrac{y - y_2}{y_m - y_2} & if\ y_2 \le y \le y_m \\ \dfrac{y_1 - y}{y_1 - y_m} & if\ y_m < y \le y_m \end{cases}$

where y is the possibility distribution of the effect E derived from the mapping. The membership of y to class L, μL(y), where class L is described by a trapezoidal function such that:

$$\mu_L(y) = \begin{cases} 0 & if\ y < a\ or\ y > d \\ \dfrac{y - a}{b - a} & if\ a \le y \le b \\ \dfrac{d - y}{d - c} & if\ c \le y \le d \\ 1 & if\ b < y < c \end{cases}$$

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ECOLOGICAL CONCERN AS A FACTOR IN THE OPTIMAL ATTENUATION OF DIVERSE STRESSOR SOURCES IN A STREAM.

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ABSTRACT

The use of an objective function based on fuzzy ecological effect expectation in a genetic optimisation algorithm to obtain site or situation specific stressor attenuation values for the management of diverse stressors emanating from several sources, is investigated. The approach is based on the premise that both regulator and regulatee are able to formulate their goals in fuzzy terms. In the case of the regulator the goals will be formulated in terms of acceptability of levels of ecological concern (a fuzzy analogy to ecological risk). In the case of the regulatee it will be formulated in terms of acceptability of the level of attenuation, which is also the control variable. A hypothetical catchment is used to illustrate the principle.

KEYWORDS
Fuzzy risk; genetic algorithm optimisation; impact assessment; impact management

INTRODUCTION

Sustainability of the aquatic environment is a water resource management goal required by many countries including South Africa. Common water resource management problems in the attainment of this goal, exacerbated in a relatively poor, water-scarce country, include:

1) integrating the impact of diverse stressors which result in the same high level effect such as loss of sustainability,
2) setting goal-related management objectives for such stressors, recognising technological or technology related constraints, and
3) the need for an equitable and transparent apportionment of impact reduction among the users of the resource.

A potential conflict between the regulatory agency charged with the protection of the resource and the users intent on using the river capacity to the full, results in pitting an apparently ethereal concept against material realities. The second and third problems are typically addressed by waste load allocation (when stressor specific numeric criteria are available) and multiple objective optimisation, both of which may entail a stochastic approach (Lohani and Thanh, 1978; Burn and McBean, 1985; Chadderton and Kropp, 1985; Burn and Lence, 1992; Hutcheson, 1992; Tung, 1992; Cardwell and Ellis, 1993; Lung, 1995) or a fuzzy approach (Hathhorn and Tung, 1989; Sasikumar and Mujumdar, 1997). The two major components that add to the stochasticity have been considered to be the variability in river and effluent flow. However, resource management to ecological goals is further complicated by:

- variability of susceptibility to stressors within and among various levels of ecological organisation,
- uncertainty introduced by insufficient system specific knowledge and,

- vagueness relating to various ecosystem-level characteristics such as integrity and sustainability (Karr and Dudley, 1981;Cairns and Niederlehner, 1995; Karr, 1996; Ludwig, et al., 1997; USEPA, 1997).

BACKGROUND

It could be argued that the main problems in solving the problem of the apportionment of impact abatement relates to a) the expression of the aggregated impact of diverse stressors and b) a formulation of the optimisation problem that is based on a common objective for resource protector and user.

The diverse-stressor problem

An expression of risk, ρ, is proposed which is epistemic of the likelihood that the system will succumb to the end-point E: loss of sustainability. Loss of sustainability is here viewed as a fuzzy end-point, which in most real cases may only characterised by qualitative, possibly vague, descriptors. As such, ρ expresses an assessment, based on available evidence, that this end-point will be attained through any stressor. This likelihood is dependent on the likelihood of the occurrence of the stressors and the likelihood of the E conditioned on the magnitude of stressors. For example, assume stressor values corresponding to E for flow-related stress, toxic substance-related stress and habitat-related stress are grouped in sets Q, T and H respectively. If the likelihood is expressed in terms of possibility, then ρ could be expressed as Eq. [1], where $\Pi(.)$ denotes a possibility measure. This possibilistic analogue of ecological risk is here referred to as ecological concern.

$$\rho = \Pi(T \cup H \cup Q) = t\text{-}conorm\{\mu_T(t),\ \mu_H(h),\ \mu_Q(q)\}. \qquad [1]$$

The right hand side of Eq. [1] is derived by considering a toxicity value t, a habitat degradation value h and a flow stress value q (with set membership functions $\mu_T(t)$, $\mu_H(h)$ and $\mu_Q(q)$ respectively), occurring in the system. The possibility that sustainability will be lost due to this set of circumstances will be expressed by ρ. In this study the *max* operator had been used to express the *t-conorm*, but a number of other operators (including the probabilistic sum) are available to tailor the operation to the situation being modelled (Klir and Yuan, 1995). The membership stressor value x to fuzzy set X has been estimated from:

$$\mu_\Phi(\phi) = \min_{\phi}\{\pi_{E|\varphi}(\phi), \pi(\phi)\} \qquad [2]$$

where $\Phi \in \{T, H, Q\}$, $\phi \in \{t, h, q\}$ and $\pi_{E|\phi}(\phi)$ and $\pi(\phi)$ are the possibility distribution of loss of sustainability conditioned on the stressor value ϕ and the possibility distribution of the stressor ϕ respectively.

Combining Eqs. [1] and [2] yields the well-known max-min composition of possibility theory (DuBois and Prade, 1994; Klir and Yuan, 1995).

$$r = \max\left\{\min_{\phi}\{\pi_{E|\phi}(\phi), \pi(\phi)\}\right\} \qquad [3]$$

Ecological concern as used here expresses the maximal expected possibility of a vague end-point (i.e. the loss of sustainability in this case). At the ecosystem level, where specific information is often sparse, expert opinion may be needed to establish, not only at what point sustainability is considered to be lost, but also to formulate the stressor response relationship. It may not be possible to stipulate any more than an expected no-effect or threshold of effect level and an expected unacceptable effect level.

Formulating the optimisation problem

The common ground between regulator and user may be found in the level of satisfaction, λ with the regulated situation. The objective for optimisation may be expressed as:

$$\begin{aligned} &\text{Max } \lambda \\ &\text{s.t.} \quad \lambda = \begin{cases} 0 & \text{if } \lambda_R < \zeta \\ \min\{\lambda_R, \lambda_x, \lambda_{eq}\} & \text{if } \lambda_R \geq \zeta \end{cases} \\ &\quad x_{ij} \geq 0 \end{aligned} \qquad [4]$$

and λ_R, λ_x and λ_{eq} are defined below.

Consider the situation where stressor i ($i \in \{1,.., n\}$) is introduced at j ($j \in \{1, .., m\}$) different points. On the part of the regulator λ will be determined by satisfaction of the management objective: $\rho \leq \rho'$ where ρ' is the concern (or risk) objective for the water body. While it would be ideal to have a crisp value for ρ', it might also be a fuzzy number not necessarily symmetric around ρ'. The concern objective may be described by ρ^{min} and ρ^{max}, levels below which the concern is perfectly acceptable and above which it is completely unacceptable respectively. The overall satisfaction with respect to the concern objective is indicated by λ_R. The values of ρ^{min} and ρ^{max} may be, in general, reach specific. Downstream of each point j, there may in principle be a different degree, $\lambda_{r,j}$, to which the concern objective $(\rho_j^{max}, \rho_j^{min})$ is satisfied. The level of concern, r_j, is the source specific concern calculated from Eq. [3] and $\lambda_{r,j}$ would a fuzzy set of Type 2 (Figure 1) on r_j and ρ_j^{min} and ρ_j^{max} as the minimum and maximum criteria respectively. As a matter of policy, it might be decided by the regulatory authority that a minimum concern satisfaction level ζ may be imposed (i.e. if the ecological concern exceeds ζ, λ =0 irrespective of other considerations). For this study $\zeta = 0$ was assumed.

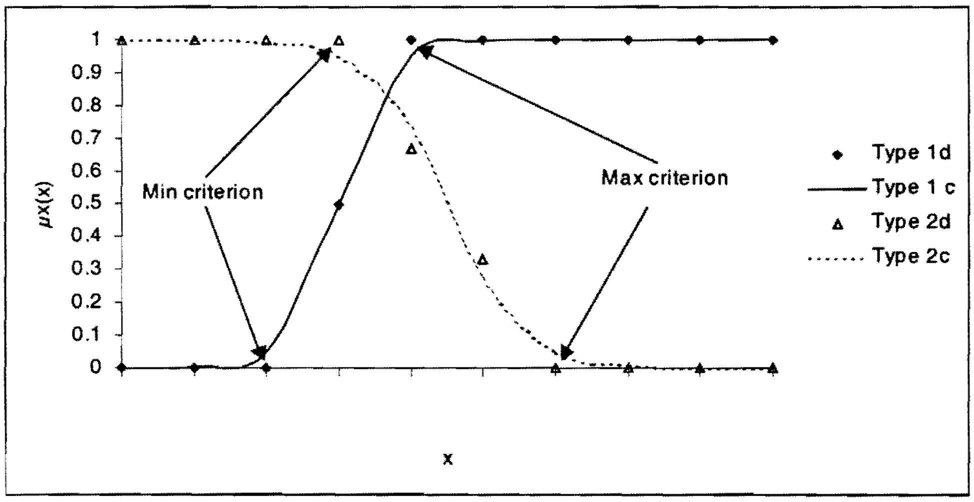

Figure 1. An illustration of the two types of fuzzy set membership functions used in this study. The d(iscrete) and c(ontinuous) versions are shown.

On the part of the regulatee, λ will be determined by stressor source management issues, specifically the acceptability of *stressor reduction* criteria for stressor sources. It is assumed that the control variable is the stressor attenuation level x_{ij} for stressor i from source j ($x_{ij} \in [0, 1]$, where no reduction implies $x_{ij} = 0$ and complete stressor removal implies $x_{ij} = 1$) for stressor i from source j. A stressor reduction level $x_{ij} > 0$ imposes a burden on the source management agency (which may be in the form of the treatment cost or some other direct or indirect operational problem). The satisfaction of each stressor source combination is indicated by λ_{ij}.

It is assumed that a crisp attenuation acceptability criterion would not be feasible and that the fuzzy equivalent can be formulated as a fuzzy set from an acceptability pair $\{x_{ij}^{min}, x_{ij}^{max}\}$ from each stressor source manager. These acceptability criteria may incorporate source- and stressor-specific weighting of cost and technological implications of a treatment level x_{ij}. Here, x_{ij}^{min} represents a treatment level that is completely acceptable, while x_{ij}^{max} represents a treatment level which, for whatever reason, is completely unacceptable. The value of λ_{ij} is a fuzzy set of type 2 (Figure 1) on xij with , x_{ij}^{min} and x_{ij}^{max} the minimum and maximum criteria respectively.

Conservatively, the value for the overall satisfaction with the regulated attenuation can be expressed as: $\lambda_x = \min\{\lambda_{ij}\}$.

The satisfaction of the requirement for equity in stressor attenuation among identical stressors is expressed as λ_{eq} is expressed as a fuzzy set of Type 2 (Figure 1) on the maximum difference (δ) in required attenuation among all stressors and sources (Eq. [5]).

$$\delta = \max_{i}^{m}\left(\frac{\max_{i}^{n}\{x_{ij}\} - \min_{i}^{n}\{x_{ij}\}}{(\max_{i}^{n}\{x_{ij}\} + \min_{i}^{n}\{x_{ij}\})/2}\right) \qquad [5]$$

Equity acceptability criterion values ε^{min} and ε^{max} of 0 and 0.2 respectively were used in this study.

METHOD

The application of this methodology is illustrated by a typical data set from a small stream in South Africa receiving water from small sewage treatment works while serving as irrigation water for smaller farms. Many such streams are at the headwaters of, or serve as refugia for major rivers. The stream is modelled as a set of four effluents and one abstraction (Figure 2). The river habitat characteristics downstream is associated with the node just upstream as part of the characteristics determining its ecological concern.

Both the stressor distribution and the site-specific conditional response ($\pi_{E|\phi}(\phi)$) are determined by expert input. The most difficult would seem to be the estimation of effect conditioned on stressor value. This is conceptually equivalent to a stressor-response relationship where the response is the epistemic possibility of observing the target effect. In all cases a minimum and maximum effect criterion (ϕ^{min} and ϕ^{max}) were elicited such that $\pi_{E|\phi}(\phi) = 0\ \forall\ \phi \le \phi^{min}$ and $\pi_{E|\phi}(\phi) = 1\ \forall\ \phi \ge \phi^{max}$. All possibility distributions were then converted to continuous function of Type 1 (Figure 1) by Eq. [6a]

The toxic substance concentrations is expressed in terms of toxicity units which in this case had been derived from an extended chronic whole effluent laboratory toxicity assessment and population growth projection (e.g. Jooste and Thirion, 1999). Based on what is known about the biota in a stream section between nodes, as well as the relative sensitivity of the laboratory test organism compared to those biota, an assessment is made of the maximum and minimum toxicity effect criteria. The toxic substance has been assumed to be subject to pseudo first-order degradation kinetics (constant 0.2 day^{-1}) and dilution. The concentrations were calculated by simple mass balance based on interval arithmetic using α-cuts from the toxic substance- and flow possibility distributions ($\alpha = 0.05$). The parameters are shown in Table 1.

The flow-response relationship is estimated from querying experts to supply q^{min} and q^{max} while using some form of instream flow requirement methodology (e.g. King and Louw, 1998). The habitat related stress-response relationship is derived similarly. The stressor exposure possibility distributions were derived directly from the corresponding probability distribution by requiring that $\max(\pi(\phi)) = 1$.

Three scenarios are presented. The parameter values for scenario A are presented in Table 1. Reach independent concern acceptability criteria of 0.05 and 0.15 ρ^{min} and ρ^{max} were used. For scenario's B and C the toxics attenuation acceptability criteria for source 1 and concern acceptability criteria respectively were changed as shown in Table 2.

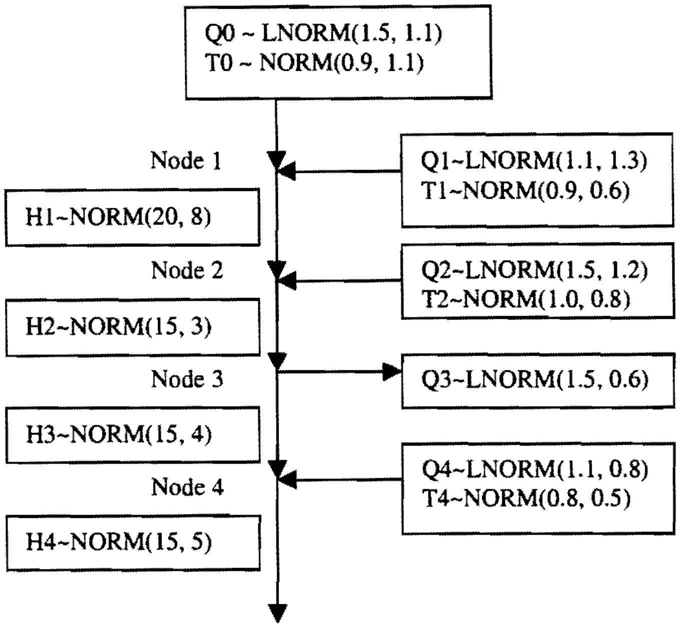

Figure 2. Schematic of the test input data used to illustrate the application of ecological concern-based optimisation. NORM (*a*, *b*) and LNORM(*a*, *b*) indicates normal and lognormal distributions respectively with median *a* and standard deviation *b*. The units for flow distributions (*Qj*) are megalitres per day, toxics distributions (*Tj*) are toxicity units and habitat degradation (*Hj*) are percent.

Table 1 Numerical input values for scenario A. (1 ML.day^{-1} = 0.0116m^3.s^{-1}.)

Parameter	Units	Node 1	Node 2	Node 3	Node 4
t^{min}	TUc	1.5	1.5	2	2
t^{max}	TUc	2.1	2.1	2.5	2.5
Flow stress effect min	ML.day-1	2.2	2.5	2.5	2.6
Flow stress effect max	ML.day-1	3.5	3.5	3.5	4
Habitat stress effect min	%	30	30	30	30
Habitat stress effect max	%	75	75	65	75
Retention time source to source	Days	2	3	2.5	4
x_q^{min}	-	-	-	0	-
x_q^{max}	-	-	-	0.6	-
x_t^{min}	-	0.2	0.2	-	0.3
x_t^{max}	-	0.7	0.8	-	0.75
x_h^{min}	-	0	0	0	0
x_h^{max}	-	0.15	0.1	0.1	0.2

Table 2. The changes in parameters associated with scenarios B and C

Scenario	$x_{t1}^{min}, x_{t1}^{max}$	ρ^{min}, ρ^{max}
B	0.01, 0.3	0.05, 0.15 (same as A)
C	0.2, 0.7 (same as A)	0.01, 0.05

The optimisation was performed using a genetic algorithm (Bäck, 1996) with search heuristics and focussing of search domain described in Ndiritu and Daniell (1999). A population of 20 solutions was used including the best four individuals from the previous generation, random crossover and a mutation rate of 0.01. The parents were selected randomly from an exponential probability distribution. After an epoch of 40 generations, 18 of the population were regenerated from an exponential distribution centred on the focussed search domain. A cycle of 10 epochs was repeated 10 times. In order to circumvent the problem of degeneracy of solutions in the optimisation heuristic, both effect and stressor distributions were modelled as the continuous approximations of the discrete sets (Figure 1). Type 1 and Type 2 continuous sets were expressed by either of Eqs. [6a] or [6b].

$$f(x) = \frac{1}{1 + a \cdot e^{-kx}} \qquad [6a]$$

$$f(x) = \frac{a \cdot e^{-kx}}{1 + a \cdot e^{-kx}} \qquad [6b]$$

Where the parameters a and k were calculated by considering the minimum criterion as corresponding to 0.05 (or 0.95 for type 2) and the maximum criterion corresponding to 0.95 (or 0.05 for Type 2).

RESULTS AND DISCUSSION

The acceptability levels are quite low (Fig. 3) when attenuation equity is required among stressor sources. The tightening of concern bounds (Scenario C, Fig. 3) results in higher attenuation levels for toxics and much lower satisfaction levels. Lowering the acceptability bounds for toxic attenuation for source 1 (Scenario B) has very little effect except to lower λ since the equity constraint tends to treat all toxics sources the same.

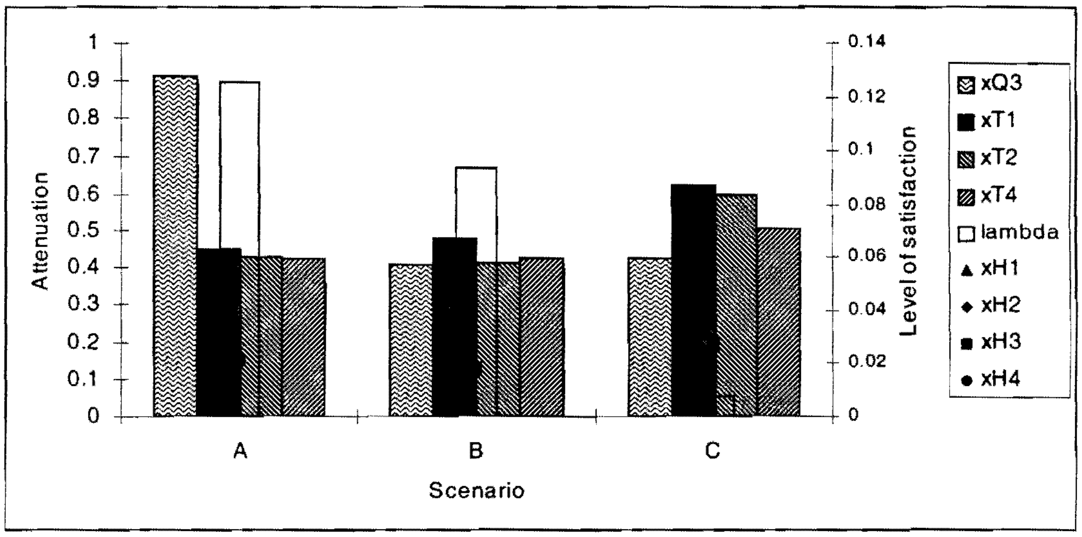

Figure 3. The attenuation levels (x) for flow (Q), toxics (T) and habitat (H) related stressors (for each of the 4 sources in the example) corresponding to the highest value of the overall acceptability λ. The value of λ is represented by the open rectangle and refers to the right hand ordinate axis.

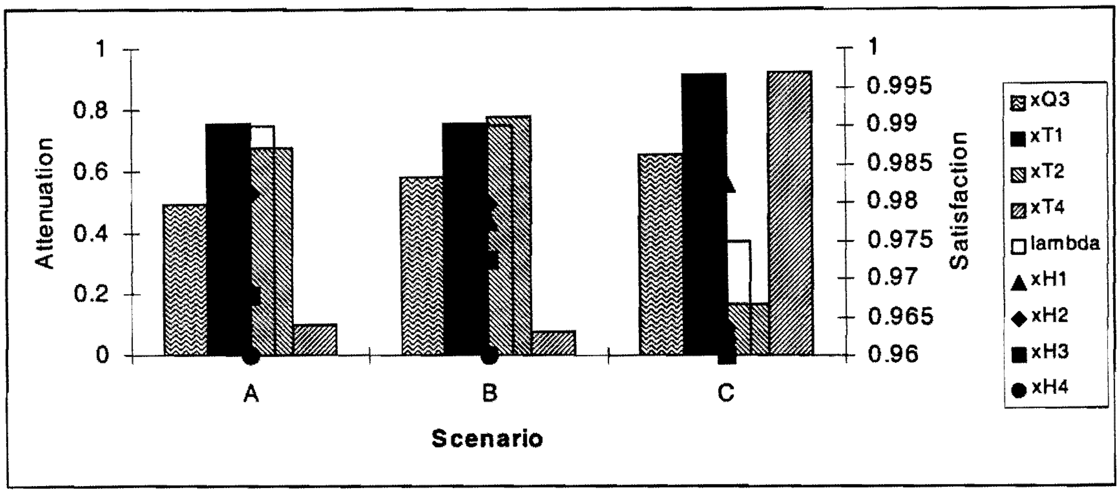

Figure 4. The effect of the removal of the equity constraint on the overall satisfaction λ and the attenuation levels (x) for flow (Q), toxics (T) and habitat (H) related stressors (for each of the 4 sources in the example) corresponding to the highest value of the overall acceptability λ. The value of λ is represented by the open rectangle and refers to the right hand ordinate axis.

If the equity constraint is removed (Fig. 4), much higher overall acceptability levels are reached. As expected, there is now also a much higher variability in stressor attenuation levels. Tightening the concern (risk) bounds highlights the more important contributors to ecological concern. In this case, habitat degradation downstream of node 4 with some contribution from toxics at node1, are probably the main contributions. It is interesting to note that flow is lower when the equity constraint is removed.

As could be expected from the values in Tables 1 and 2, toxic emission attenuation impacts the most on the ecological concern values and consequently demands the highest attenuation. However, while it would normally have been expected that sources 1 and 2 would require the highest attenuation (Figure 4), equity considerations lowers the attenuation for these sources at the cost of increasing attenuation at source 4 (Figure 3). The feasibility of doing this would obviously depend on local conditions. Although the true optimum may not have been reached on the imposition of equity constraints, it would seem likely that the abstraction attenuation would be higher compared to the situation where equity is not required. This would be the result of the greater weight accorded to the larger number of sources: a larger number of abstractors in the system would have evened out this effect. In the South African situation, for example, given the relative scarcity of water and the dependence of agriculture on irrigation, equity constraints may well have to be waived. This would clearly be a matter of negotiation or policy.

Other results (Jooste 2000) confirmed that λ_x tends to dominate the overall acceptability of the solutions and that λ_R and λ_{eq} tended to be much higher than λ_x. While ecological concern considerations would appear to raise the attenuation values, the source- and stressor specific acceptability consideration are still limiting. The implication here is that, unless the factors determining attenuation acceptability criteria are addressed, no further impact reduction could be expected. Since these factors may include both economic and technological considerations, addressing them may also have far reaching ramifications.

These results and the assumptions on which they were based would have definite policy and catchment management implications. However, the results in themselves may serve as a useful tool in decision making, supplying at least a baseline for decision making with a view to ecological protection.

CONCLUSION

Ecological concern, like ecological risk, makes use of available data on both the occurrence of stressors and the expected effect of these stressors. The likelihood nature of ecological concern lends itself to the aggregation of the contribution of diverse stressors if a common effect (such as loss of sustainability) is chosen as an end-point. However, it requires an explicit statement of at least semi-quantitative concern (or risk) objectives.

The ecological concern approach to stressor management may prove to be a useful tool in water resource management policy formulation as well as situation analysis under conditions where ecological goals need to be integrated with point source management issues. Although the information requirement for this approach is not insignificant, it provides a platform on which water quality and quantity issues can be integrated. It may be a basis on which stressor and source specific criteria can be generated. The practicality of this methodology would be influenced by a) the knowledge base available to estimate the conditional effect possibility, and b) the spirit of co-operation among the regulator and the stressor-source manager.

It is recognised that the estimation of the conditional effect possibility and the stressor attenuation acceptability criteria as described here, is essentially subjective. This process needs to be formalised and refined possibly drawing on the extensive work done on fuzzy expert systems. The formulation of objective procedures to derive these critical parameters will certainly facilitate the use of ecological concern as a water resource management tool.

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CHAPTER 5

CONCLUSION: APPLICATION AND THE WAY FORWARD

5.1 ABSTRACT

This chapter describes three possible applications for the models developed in preceding chapters as tools in resource directed catchment management:

- Rapid hazard ranking in rapid reserve estimates,
- The derivation of in-stream stressor specific criteria, and
- The derivation of baseline point-source criteria in catchments under development pressure.

Some of the necessary work that needs to be performed to place risk-related catchment management on a sound scientific basis and incorporate it in the current water resource management practice include:

- Development of a policy on risk assessment and risk management
- Deriving risk objectives
- Establishing a risk communication policy
- Investigating more efficient optimisation algorithms
- Deriving and updating stressor response relationships (SRR's)
- Development of rigorous methodology for the characterising stressor attenuation acceptability.

5.2 IN SUMMARY

In Chapter 1 a potential Reserve-related problem in dealing with diverse stressors from multiple sources against the background of the mandatory protection of a sustainable aquatic ecosystem. It was proposed that it be by risk methodology. A broad outline of how ecological risk-based management (ERBM) might be applied, was given.

In Chapter 2 the ecological risk assessment (ERA) methodology was outlined and how it needs to be adapted for ERBM. Considering its theoretical background it was clear that the problem of projecting end-points from laboratory scale data to the ecosystem level involves a large amount of uncertainty since it requires not only scale projection but also conceptual projection. This process needs to be performed for each individual stressor risk. The end-point projection forms a very important task in the construction of SRR's.

Chapter 3 modelled the aggregate risk of diverse stressors as the disjunction of individual stressor risk. It was illustrated how this type of aggregation could be used in both a probabilistic and possibilistic framework.

Chapter 4 modelled the diverse-stressor-multiple-source problem as an optimisation problem. A genetic algorithm was chosen to solve the optimisation problem, not because it is necessarily the most efficient, but because it is conceptionally simple. It was illustrated that it is indeed possible to obtain source- and stressor-specific attenuation criteria.

5.3 A PERSPECTIVE ON THE WORK PRESENTED

It became clear that the inputs needed to make these procedures functional are quite information- and knowledge-intensive. Even though the necessary knowledge exists, the risk-based decision-making is unlikely to be a first choice approach unless the stakes are high enough to warrant the time and effort to generate the necessary data.

5.3.1 INFORMATION NEEDS

The scientific input to risk methods largely comprises of uncertainty, variability and vagueness characterisation as well as risk characterisation. All of these depend strongly on insight into the functioning of the aquatic ecosystems, as expressed in conceptual models of various kinds, the type and quality of data available and the experience and insights of the body of experts available in the country. When moving towards the benefits of using risk-based methodology, it should be recognised that the quality of scientific basis of risk-techniques need to be carefully considered and expertise in a small country, like South Africa, needs to be nurtured.

5.3.2 ACCOMODATING UNCERTIANTY

ADJUSTMENT IN THE REGULATORY PARADIGM

The regulatory mechanisms need to be adjusted to (a) recognise that uncertainty (in it's broadest sense) is a fact of life in ecosystems management and (b) that rather than to try to define the uncertainty out of the process, incorporate the methodology to deal with it in the process. A vast literature exists in the area of business and engineering decision-making under uncertainty (see for example Chapter 1 in Stewart, *et al*, 1997), so that uncertainty need not be seen as a bane to regulatory decision-making.

RISK COMMUNICATION NEED

By nature, human beings have a fear of the unknown and of uncertainty. Innately, therefore, when a decision is made in an area of which they do not have knowledge and by mechanisms they do not understand, people tend to be distrustful. If, in addition, they suspect that the motives behind the decision are suspicious or antagonistic to their value system, distrust may turn to hostility.

Suspicion of the scientific domain may lead to remembering catastrophes of the past, such as the thalidomide scandal of the sixties and the uncontrolled use of DDT in the 1950's. The use of risk by the scientific community and particularly in the industrial context has been seen as an excuse for doing nothing (Tal, 1997). These issues need to be addressed by effective risk communication, which is generally recognised as an increasingly important aspect of risk application (CRARM, 1997; OECD, 1997; Yosie, et al., 1998).

5.4 POSSIBLE APPLICATIONS OF THE METHODOLOGY DESCRIBED IN THIS STUDY

Risk may reasonably be used to aid water resource quality management decisions and activities related, but not necessarily limited to the following areas:

5.4.1 BASIS FOR STRESSOR-SPECIFIC WATER QUALITY CRITERIA.

The current South African Water Quality Guidelines for the Aquatic Environment has been derived from toxicological data (typically concentration-response data) and some qualitative assumptions regarding exposure. These criteria have the following limitations:

- the derivation process produces anomalous risk results so that the expected effect differs from substance to substance,
- recognition could not be given to the co-occurrence of different stressors since they could not be expressed on a common basis, and
- the criteria do not necessarily relate to the same ecological effect.

Redefining and recalculating the criteria on a risk basis induces a measure of transparency into the interpretation of the criteria. The other criteria (besides those for the aquatic environment) could be approached similarly. Both ERA and human health risk assessment will be important here. Methodologies have been developed for the determination of the ecological reserve. These methodologies follow relatively independent routes to establish stressor-specific management criteria. These criteria characterise the reserve for a particular river reach. In the form these criteria are currently expressed, there is no description of the uncertainty component in the relationship between the stressor and its effect. It is likely that the various stressor criteria project to different risk levels. A significant improvement in the homogeneity of the process can be brought about by:

1. describing the management classes in risk terms
2. adopting suitable numeric risk objectives
3. deriving SRR's for effect likelihood at the statutory end-point for all identified stressors
4. adopting numeric risk objectives which are related to the management goal
5. calculating the corresponding stressor exposure-likelihood level and hence the management criteria for the designated stressors by iterative application of the models in Chapter 3.

Each of the steps 3 and 5 above can be performed at various levels of environmental realism, ranging from a highly simplified desktop estimate, which is a rapid, low confidence, estimate to a moderately long term, high confidence site-specific study.

In its simplest form this procedure would involve:

(a) Assuming a type distribution for the stressor (e.g. a lognormal distribution).

(b) Iteratively adjusting the location and scale parameters of the distribution and comparing the calculated risk from each parameter vector with the risk objective. This would call for optimisation and may involve two dimensional uncertainty analyses.

(c) Describing benchmarks of the stressor distribution (e.g. median and 95th percentile).

It is clear that the quality of the SRR is vitally important.

5.4.2 THE DERIVATION OF BASELINE POINT-SOURCE CRITERIA IN CATCHMENTS UNDER DEVELOPMENT PRESSURE.

The issue of diverse stressor-multiple source management under constraints was the main focus of this study. The technical process of the multiple-source problem is described in Chapter 4. The diverse-stressor problem formulation requires some extra information. The first four steps of 5.4.1 is followed, but the following steps are added:

5. define catchments or river reaches subject to development pressure,
6. obtain source- and stressor-specific upper and lower limits of stressor attenuation from stressor sources with particular attention to the uncertainty in these estimates,
7. define, either as a matter of policy, or pragmatically, the relative weighting of source and regulator satisfaction,
8. estimate the source attenuation terms along with its confidence estimates, and
9. finalise the management criteria by negotiation between regulator and regulatee(s) based on attenuation estimates.

The derivation of the stressor-source specific attenuation must be followed by a calculation of the actual stressor values represented by the level of attenuation. This could then be compared to the source criteria derived from WLA for example (in the case of substance stressors). In evaluating the implications of different Hazard- or risk-based in-stream stressor criteria and the criteria derived in terms the DSMS solution it should be remembered that:

- the DSMS criteria are risk based and therefore not comparable to hazard-based criteria
- the DSMS criteria are derived from catchment considerations and do not address site-specific considerations.

If the DSMS stressor criteria are more lenient than the other criteria, the DSMS criteria might serve as the short-term criteria but with the proviso that whichever constraints hamper the achievement of the other criteria should be resolved on the longer tem. If the converse is true, the stricter of the two should be used.

5.4.3 RESOURCE MANAGEMENT CLASSIFICATION.

The provision in the National Water Act for the classification of water resources can reasonably be linked to risk concepts. Management objectives may more specifically be expressed in terms of allowable risk to the Reserve. This provides an explicit communality between the receiving water quality/risk objectives and the Reserve as well as effluent criteria and/or standards.

5.4.4 HAZARD RANKING.

In some situations, it is neither necessary nor feasible to calculate absolute risks. In the case where different hazards within the same scenario or hazards in different scenarios need to be compared, risk is often a suitable basis for comparison. The management criteria derived in the current reserve determinations (McKay, 1999) are largely hazard based. Realistic ranking of the hazards addressed in this process can be accomplished by estimating the risk attached to these hazards. This would require:

⇨ a clear statement of a realistic worst case stressor exposure scenario,

⇨ a clear conceptual ecological model linking the level of data with the required end-point,

⇨ an expression of the uncertainty in the SRR, and

⇨ an estimate of the risk.

This will aid in characterising the uncertainty and channelling expenditure into areas of greatest return.

5.5 ISSUES IN THE APPLICATION OF RISK METHODOLOGY

The major areas where attention needs to be given to give effect to risk-based catchment management are:

5.5.1 THE DEVELOPMENT OF A POLICY ON RISK ASSESSMENT AND MANAGEMENT

Some aspects involved in a policy on risk and risk assessment include:

- A common understanding of the definition of risk.
- How risk is seen in relation to other paradigms.
- What conditions might indicate the use of risk methodology
- Adoption of a tiered approach to the use of risk as an assessment technique
- Minimum requirements for risk assessment.

An analysis of the regulatory situation in other countries (Table 5.1) shows that the lack of a legal basis for the explicit use of risk methodology in South Africa is not unique. The National Water Act (like many other laws in South Africa) allow for the promulgation of regulations under the Act and application of risk may well be described in such regulation.

Table 5.1 An assessment of legal standing of risk assessment in selected countries (based on OECD, 1997).

Country	Law prescriptive/ goal setting	Risk criteria identified/ specified	Quantified risk assessment recognised
Germany	Prescriptive	No	No
France	Some prescriptive	Yes (zoning)	No
Switzerland	Both	In guidelines	Yes
UK	Both (more goals)	In guidelines	Yes
USA	Goal	Specific goals and definitions	No (can be used)
Norway	Goal (by industry)	No	Yes (implicitly)
Netherlands	Goal	Yes (not in law yet)	Yes

The OECD (1997) notes a potential legal problem in explicitly incorporating risk in laws since it may be asked whether generating and accepting a measure of risk will infringe the rights of individuals. This will clearly have to be assessed on a country-specific basis.

5.5.2 DEVELOPING RISK OBJECTIVES

In the foregoing work, it had been implicitly accepted that recognised risk criterion values are available, whether crisp or fuzzy. Such values for aquatic ecosystems are rare if existing at all. The reason, most likely, is that consensus on the actual numeric value as well as the descriptive risk, is likely to depend on the specific situation that is being assessed and factors such as the protection value of the ecosystem will probably have an impact. The situation with the ecological Reserve in South Africa already lends itself to a discretisation of aquatic ecosystems. An importance and sensitivity rating of river systems is being developed for river reaches (Kleynhans, 1999a), which will be factored into the Reserve determination. This could serve as a basis for ascribing maximum acceptable risk values depending on the importance class.

The decision on numeric risk criteria, i.e. what levels of probabilistic and possibilistic risk are considered acceptable, for human health considerations are generally founded on those used by the USEPA. For carcinogens a risk limit of 10^{-6} per lifetime is accepted and for non-carcinogens a value of 10^{-4} per lifetime.

For ecosystems the acceptable risk limit is likely to be more problematic. The values that will be accepted may well depend on the end-point. The risk of a major fish-kill and that of long term unsustainability may be perceived differently because the end-point relate to different time-scales. A fish kill may, because of the immediacy of effect, be rated higher than a long-term effect.

A recent study (Jooste *et al.*, 2000) considered the setting of risk objectives (RO's) by comparison with actuarial risk values. Some of the suggested values are listed in Table 5.2. These could be combined with the qualitative description in Table 5.3 to provide probabilistic risk criterion values.

Table 5.2 Human mortality risk benchmarks for establishing and communicating risk (from Chapman and Morrison, 1994)

Cause	Probability
Motor vehicle accident (USA)	1 : 100
Smoking (20/day) all effects	1 : 200
Murder	1 : 300
Fire	1 : 800
Firearm accident	1 : 2 500
Electrocution	1 : 5 000
Asteroid/ comet impact	1 : 20 000
Passenger aircraft crash	1 : 20 000
Flood	1 : 30 000
Tornado	1 : 60 000
Venomous bite/ sting	1 : 100 000
Fireworks accident	1 : 1 000 000
Food poisoning (botulism)	1 : 3 000 000
Drinking water with EPA limit of trichloro-ethylene	1 : 10 000 000

Table 5.3 A semi-quantitative approach to risk characterisation

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

The occurrence of some of the ecological events described in Table 5.2 may be difficult to define. It may, for example, be argued that smoking constitutes a generally acknowledged high risk activity and that, therefore, the highest risk that will be allowed for a chosen significant end-point will also be 1: 200. On the other hand, flying in a passenger aircraft is generally considered safe and that, therefore, a risk of 1: 20 000 may be considered negligible. These values would likely be determined on a case specific basis.

5.5.3 RISK COMMUNICATION

In the catchment management situation, which is also the likely setting for the diverse-stressor-multiple-source problem, it could be envisaged that communicating and defending the risk criteria selected for a river reach would arise. This requires dealing with the sociological problem of risk perception. Perceptions about risk change with changing circumstance and increasing familiarity; increased familiarity with a hazard leads to a better estimate of its true probability of occurrence, or conversely, the more unfamiliar one is with a hazard, the more one is inclined to overestimate the danger (OECD, 1997; Tal, 1997). The way in which risks are communicated in a tense situation, could have a significant impact on the viability of the methodology described in Chapter 4 particularly.

5.5.4 INSITUTIONALISING RISK MANAGEMENT IN THE RESERVE CONTEXT

There needs to be a formal awareness of uncertainty in ecological management. This would involve an institutional concern with the variability, uncertainty and vagueness pertaining to the ecosystem and an insistence on all management levels of explicitly stating or asking for such expressions, in order to contextualise management decisions. This would involve:

- Developing a generic "first attempt" ecological model for risk assessment.
- Cultivating an institutional awareness of SRR's and their importance in effect driven management
- Creating risk-susceptible administrative procedures e.g. risk oriented discharge permits
- Developing risk assessment capacity
- Developing risk communication capabilities

5.6 RESEARCH NEEDS: THE WAY FORWARD

The work presented in this study on the derivation of effect-likelihood criteria in a diverse-stressor multiple-source (DSMS) management situation, addressed an aspect of ERA that had not received much attention in the past. Some of the issues addressed in this study require a multi-disciplinary or trans-disciplinary approach, which increased the difficulty of the task significantly. Some of the issues were, consequently, left unresolved although they may be quite significant. Some of the more significant problems that would still need to solved include:

1) Investigating the use of other optimisation algorithms, e.g. simulated annealing and stochastic optimisation methods. The genetic algorithm that was used in the DSMS problem solution,

although sufficient for the small number of control variables in the illustrative situation used, may not work as well in a higher dimensional space.

2) Deriving stressor-response relationships for all common stressors to reserve related end-points. The possibilistic approach used in Chapters 3 and 4 may not suffice in situations where higher precision values are necessary. The probabilistic analogue to this approach needs to be researched.
3) Establish formal feedback loops between SRR's and instream bio monitoring to inform and improve both the SRR's and the biomonitoring programme design. Once again, the possibilistic Dempster-Schafer approach using possibility distributions has to be extended to the probabilistic analogue. This may involve investigating the use of Bayesian methodology.
4) Improving the stressor modelling sophistication of the model in Chapter 4. The Possibilistic approach was chosen because it appeared that the data were better suited to the situation. Both the stochastic approach and a more sophisticated environmental model could be used to improve the realism of the stressor value prediction in suitable situations.
5) Developing methods to characterise source attenuation acceptability in a rigorous manner. The assumption in Chapter 4 had been that suitable methodologies exist by which stressor source managers could estimate the acceptability of attenuation values. It is not immediately apparent that these methods already exist and some effort might well required to formulate credible, transparent methodology to define such acceptability values rigorously.

APPENDIX TO CHAPTER 1

A1.1 A REVIEW OF SOME PERTINENT ASPECTS OF THE SOUTH AFRICAN NATIONAL WATER ACT (ACT 36 OF 1998).

The aim of this study is to provide a tool to be used in water resource management with a view to the protection of the aquatic ecological Reserve as defined in the National Water Act in South Africa. While the application of the approach may be much wider than the aquatic system, this study must be seen against this backdrop.

In its preamble, the rationale for the Act comes from recognising that:
(a) "water is a scarce and unevenly distributed resource",
(b) "the ultimate aim of water resource management is to achieve the sustainable use of water for the benefit of all users",
(c) "the protection of the quality of water resources is necessary to ensure sustainability of the nation's water resources" and
(d) there is a "need for the integrated management of all aspects of water resources".

Section 2 of the Act states that 'the purpose of this Act is to ensure that the nation's water resources are protected, used, developed, conserved, managed and controlled in ways which take into account amongst other factors-
(a) meeting the basic human needs of present and future generations;
(b) promoting equitable access to water;
(c) ...
(d) promoting the efficient, sustainable and beneficial use of water in the public interest;
(e) facilitating social and economic development;
(f) ...;
(g) protecting aquatic and associated ecosystems and their biological diversity
(h) reducing and preventing pollution and degradation of water resources; ...

Some of the pertinent definitions that will be used here will be used in a manner similar that in the Act:

" ....

(iii) '**catchment**' in relation to a water course .... means the area from which any rainfall will drain into the watercourse.... Through surface flow to a common point or points.

(xi) '**in stream habitat**' includes the physical structure of the watercourse and the associated vegetation in relation to the bed of the watercourse;

(xv) '**pollution**' means the direct or indirect alteration of the physical, chemical or biological properties of the water so as to make it- ....;

(b) harmful or potentially harmful-

(aa) to the welfare health or safety of human beings;

(bb) to any aquatic or non-aquatic organisms;

(cc) to the resource quality; or ...;

(xvii) '**protection**' in relation to a water resource, means- (a) maintenance of the quality of the water resource to the extent that the water resource may be used in an ecologically sustainable way; (b) prevention of the degradation of the water resource; and (c) rehabilitation of the water resource;

(xviii) '**Reserve**' means the quantity and quality of water required-

(a) to satisfy basic human needs....; and

(b) to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource

(xix) '**resource quality**' means the quality of all aspects of a water resource, including-

(a) the quantity, pattern, timing, water level and assurance of in stream flow;

(b) the water quality , including the physical, chemical and biological characteristics of the water;

(c) the characteristics and condition of the in stream and riparian habitat; and

(d) the characteristics, condition and distribution of aquatic biota;

(xxii) '**waste**' includes any...material that is suspended, dissolved or transported in water (including sediment) and which is ... deposited ... into a water resource in such volume, composition or manner as to cause ... the water resource to be polluted;

(xxiv) '**watercourse**' means ... a river ... [or] a natural channel in which water flows regularly or intermittently ... and ... includes, where relevant, its bed and banks;

(xxvii) '**water resource**' includes [*inter alia*] watercourse [and] surface water.

......".

Section 6 of the Acts requires that the water resource strategy (which may be phased) should (6 (b) (i)) provide for the requirements of the Reserve and (6 (i)) state the water quality objectives for the water resource.

Sections 12 and 13 make provision for the classification of the water resource, although it does not specify the basis for classification. This classification system must also serve as the basis for setting the resource water quality objectives. The objectives may relate to:

(a) the Reserve
(b) the in stream flow
(c) the water level
(d) the presence and concentration of particular substances in water
(e) the characteristics and quality of the water resource and the in stream and riparian habitat
(f) the characteristics and distribution of aquatic biota
(g) the regulation of in stream or land-based activities
(h) any other characteristics

of the water resource.

The impact of the Reserve on water use and water management can be seen by considering that:

- Section 15 makes it mandatory that any action that follows from the Act must give effect to this class and its associated water resource quality objectives while Section 18 demands that such actions must also give effect to the Reserve. Section 16 determines that the Reserve must also be set in accordance with the class. This places the Reserve central to water resource management.

- Under Section 22. (7)(b)(i) compensation which is payable on the reduction of lawful use of water does not apply to reduction of water use to make provision for the Reserve.

- Section 56 makes provision for establishing a pricing strategy which may contain a strategy for water use charges for funding water resource management to protect the resource, including the discharge of waste and the protection of the Reserve (55.(2)(a)(iv)).

In making regulations on water use, besides giving effect to the Reserve and the resource classification system, Section 26 requires that, *inter alia*, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the in stream and riparian habitat. Water use regulation must take into account factors such as (Section 27. (1)):

1. The socio-economic impact of water use or curtailment of use (d)
2. The catchment management strategy applicable to the resource (e)
3. The likely effect of the water use on the resource and other users (f)
4. The class and resource quality objectives (g)
5. The investment already made and to be made by the water user (h)
6. The quality needs of the Reserve and to meet international obligations (j)

A1.2 RISK AND HAZARD: PARADIGMS AND STYLES

A1.2.1 THE HAZARD AND RISK ASSESSMENT PARADIGMS

Given that monitoring and assessment are essential components of any management strategy, the assessment paradigm is crucial to the expectations and format of the assessment of management goal attainment. The assessment may take the form of either a quantal or a continuous metric. The quantal assessment paradigm (QAP) and continuous assessment paradigm (CAP) are referred to as hazard and risk assessment paradigms (Figure A1.1) respectively by Suter, (1993). The characteristics of these paradigms are summarized in Table A1.1 and the progress of an assessment according to these paradigms is illustrated in Figure A1.1.

Table A.1.1. Characteristics of environmental hazard assessments and risk assessments (adapted from Suter, 1993). Some of the characteristics are explained in the text.

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

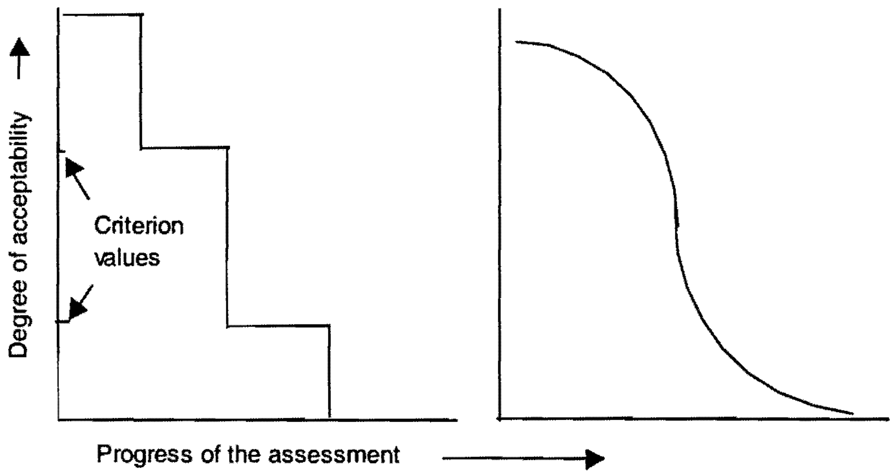


Figure A1.1 A representation of the outcome of an assessment as the assessment progresses. In the progress of the assessment, the confidence in the data increases. In this example both assessments starts out with the assumption of unacceptable for a situation that is essentially acceptable.

A comparison between the QAP and CAP paradigms reveals:

1. Both QAP and CAP assume that the environmental safety of a substance should be based on the relationship between the degree of toxicity and the extent of exposure. This differs in principle from technology-based assessment.

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

a) Reliance on scientific **judgement** or "expert opinion" of what constitutes "acceptable" or "unacceptable". The expert opinion may be either explicitly stated or encapsulated in a criterion value (CV).

b) ANOVA techniques and statistical **hypothesis testing** play an important role in the QAP in deciding whether the expected (or measured) environmental concentration (EC) differs from the CV.

c) A fundamental assumption of the QAP is that, given enough time and effort, the situation where the EC, for example, cannot be confidently fit into either category, can be resolved (i.e. it can in principle always be assigned a unique outcome). In a situation where no clear, unequivocal answer is available in assessing the status of an observation relative to the criterion, the hazard paradigm demands tiered iterative data gathering (testing and measurement) procedure until a definitive answer can be given. This gives rise to a **tiered assessment**. As more iterations are added to the process the confidence in the distinction between acceptability and unacceptability grows. Confidence here does not necessarily refer to statistical confidence, but more so to institutional or personal confidence (Suter, 1990).

d) Formally, there is not necessarily an explicit decision *ab initio* as to which end-points that are being addressed; it does not intend to identify what is specifically expected to occur (Bartell, *et al*, 1992) since these are implicit in the criteria. Both the process by which the expert selects the end-point (i.e. what might be expected to occur) and the extent to which this is possible is subjective to a degree even though it may be internally coherent. This aspect of the QAP makes the process inherently less transparent.

3. The continuous assessment paradigm (CAP) is characterised by:

a) Acceptance, *a priori*, that some uncertainties are practically irreducible and that a definite decision on yielding acceptable/unacceptable may be logically impossible. Consequently, there are decisions that may never (within the time frame of the decision making process) have a deterministic answer and therefore relies more heavily on **probabilistic expression**.

b) Accepting a **continuum** "grey scale" in assessment outcome. This results from its use of probabilistic assessment methods to accommodate uncertainty explicitly.

c) Because of its probabilistic expression, the object and **end-point appears explicitly** in the assessment (the probability of what could happen to whom).

d) In most environmental assessment situations, the risk paradigm would appear to be **more objective** means of decision-making. It must however be accepted that some form of human judgement can never be completely removed from the risk paradigm. For example, what constitutes a large or a small risk is often a matter of subjective judgement or policy.

APPENDIX TO CHAPTER 2

A2.10 ASPECTS OF END-POINT PROJECTION

A2.10.1 THE CRISP INFERENTIAL RULE BASE

The rule base deriving from the conceptual model can be stated as:

IFF *Sustainability* IS *assured* THEN *Integrity* IS *intact* **Rule I**

IF *Integrity* IS *intact* THEN (*Biodiversity* IS *adequate* AND *Temporal stress and recovery patterns* IS *largely undisturbed* AND *Biotic stress* IS *insignificant*) **Rule II**

IF *Biodiversity* IS *adequate* THEN (*Composition* IS *intact* AND *Structure* IS *intact* AND *Function* IS *normal*) **Rule III**

IFF *Composition* IS *intact* THEN NOT (*Composition stress* IS *present*) **Rule IVa**

IF (*Composition stress* IS *present*) THEN (*exposure to stressor 1* IS *present*) OR (*exposure to stressor 2* IS *present*) OR ... **Dummy Rule 1**

IF (*exposure to stressor 1* IS *present*) THEN [(*significant level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *long*)] OR [(*High level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *significant*)] **Dummy Rule 2**

Combining Dummy Rules 1 and 2:

IF (*Composition stress* IS *present*) THEN

{[(*significant level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *long*)]
OR [(*High level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *significant*)]}
OR {[(*significant level of stressor 2* IS *present*) AND (*exposure duration to stressor 2* IS *long*)]
OR [(*High level of stressor 2* IS *present*) AND (*exposure duration to stressor 2* IS *significant*)]}
OR …. **Rule Va**

(IFF denotes "if and only if")

Rules IVa and **Va** is repeated for *Structure* and *Function* to yield the equivalent rules **IVb, Vb, Vc** and **Vc** respectively. Using the key:
Sus: Sustainability is assured, *Res*: Resilience is assured, *Int* : Integrity is assured, *Div*: Biodiversity is intact, *Tpat*: Temporal stress/recovery patterns are undisturbed, *Cmp*: System composition is undisturbed, *Str*: System structure is undisturbed, *Fct*: System function is normal, *Tpats*: Temporal stress/ recovery patterns are in a state of stress, *Cmps*: System composition is under stress, *Strs*: System structure is under stress, *Fcts*: System function is under stress, *lxi0*: Minimally significant level of stressor X exists for integrity component i, *dxi0*: Minimally significant duration of exposure to stressor X exists for integrity component i, *dxi* : Long duration of exposure to stressor X exists for integrity component i, *lxi*: Intense exposure to stressor X exists for integrity component i, where $X \in$ {toxic substances (T), flow deficiency (Q), nutrient disruption (N), system driving variables disruption (S), physical habitat disruption (H)}, and $i \in$ {*Cmp* (c), *Fct* (f), *Str* (s), *Tpat* (t)}.

The rules can be translated to a canonical form with the standard logic operators ($\rightarrow$ "implies", $\leftrightarrow$ "equivalent to", $\neg$ "not" $\wedge$ "disjunction", $\vee$ "conjunction"):

Rules I	$Su \leftrightarrow Int$	[A2.1](Assumption)
Rule II	$Int \rightarrow Div \wedge Tpat \wedge B$	[A2.2]
Rule III	$Div \rightarrow Cmp \wedge Str \wedge Fct$	[A2.3]
Rule IVa	$Cmp \rightarrow \neg Cmps$	[A2.4a]
Rule IVb	$Str \rightarrow \neg Strs$	[A2.4b]
Rule IVc	$Fct \rightarrow \neg Fcts$	[A2.4c]
Rule IVd	$\mathrm{Tpat} \rightarrow \neg \mathrm{Tpats}$	[A2.4d]
Rule Va	$Cmps \rightarrow \bigcup_{x \in X} (lxc0 \wedge dxc) \vee (lxc \wedge dxc0)$	[A2.5a]
Rule Vb	$Strs \rightarrow \bigcup_{x \in X} (lxs0 \wedge dxs) \vee (lxs \wedge dxs0)$	[A2.5b]
Rule Vc	$Fcts \rightarrow \bigcup_{x \in X} (lxf0 \wedge dxf) \vee (lxf \wedge dxf0)$	[A2.5c]
Rule Vd	$Tpats \rightarrow \bigcup_{x \in X} (lxt0 \wedge dxt) \vee (lxt \wedge dxt0)$	[A2.5d]

Where $\bigcup_{x \in X} \bullet$ indicates the disjunction of $\bullet$ over all the stressors.

The implication of the assumption $Sus \leftrightarrow Int$ ([A2.1]) is that epistemologically sustainability does not differ from integrity. Consequently, the uncertainty associated with each of these is similar.

Given : $\neg(A \wedge B) = \neg A \vee \neg B$ and $\neg(A \vee B) = \neg A \wedge \neg B$ and if $A \rightarrow B$ then $\neg B \rightarrow \neg A$, equations [A2.5a] to [A2.5d] become[A2.6a] to [A2.6d] respectively,

$$\neg Cmp \rightarrow \bigcup_{x \in X} (lxc0 \wedge dxc) \vee (lxc \wedge dxc0) \quad \text{[A2.6a]}$$

$$\neg Str \rightarrow \bigcup_{x \in X} (lxs0 \wedge dxs) \vee (lxs \wedge dxs0) \quad \text{[A2.6b]}$$

$$\neg Fct \rightarrow \bigcup_{x \in X} (lxf0 \wedge dxf) \vee (lxf \wedge dxf0) \quad \text{[A2.6c]}$$

$$\neg Tpat \rightarrow \bigcup_{x \in X} (lxt0 \wedge dxt) \vee (lxt \wedge dxt0) \quad \text{[A2.6d]}$$

Combining Eqs. [A2.4a] to [A2.4c], [A2.3], [A2.2] and [A2.1] yields [A2.7], [A2.8] and [A2.9].

$$\neg(Cmp \wedge Str \wedge Fct) = \neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div \quad \text{[A2.7]}$$

$$\neg(Div \wedge Tpat) = \neg Div \vee \neg Tpat \rightarrow \neg Int \quad \text{[A2.8]}$$

$$\neg Int \leftrightarrow \neg Sus \quad \text{[A2.9]}$$

A2.10.2 FUZZY INFERENCE RULE BASE

A restatement of the crisp rules on which the inference system depend along the lines of these principles will highlight the need for a fuzzy logic approach (the ~ indicates the fuzzy formulation):

IFF *Sustainability assurance* IS *very high* THEN *Resilience assurance* IS *very high* **Rule I~**

IFF *Resilience assurance* IS *very high* THEN *Integrity maintenance* IS *very high* **Rule II~**

IF *Integrity maintenance* IS *very high* THEN (*Biodiversity* IS *normal* AND *Temporal stress and recovery patterns* IS *natural*) **Rule III~**

IF *Biodiversity* IS *normal* THEN (*Composition* IS *pristine* AND *Structure* IS *intact* AND *Function* IS *normal*) **Rule IV~**

IFF *Composition* IS *pristine* THEN NOT (*Composition stress* IS *significant*) **Rule Va~**

IF (*Composition stress* IS *significant*) THEN (*exposure to stressor 1* IS *critical*) OR (*exposure to stressor 2* IS certainly *critical*) OR ... **Dummy Rule1~**

IF (*exposure to stressor 1* IS *critical*) THEN [(*level of stressor 1* IS *marginally significant*) AND (*exposure duration to stressor 1* IS *long*)] OR [(*Level of stressor 1* IS *high*) AND (*Exposure duration to stressor 1* IS *marginally significant*)] **Dummy Rule 2~**

IF (*Composition stress* IS *significant*) THEN {[(*Level of stressor 1* IS *at least marginally significant*) AND (*Exposure duration to stressor 1* IS *long*)] OR [(*Level of stressor 1* IS *high*) AND (*Exposure duration to stressor 1* IS *at least marginally significant*]}

OR{[(*Level of stressor 2* IS *at least marginally significant*) AND (*Exposure duration to stressor 2* IS *long*)] OR [(*Level of stressor 2* IS *high*) AND (*Exposure duration to stressor 2* IS *at least marginally significant*]}

OR .... **Rule VIa~**

A2.10.3 LOWER LEVEL PHENOMENA

At this point a connection with the integrity-related variables needs to be made with the laboratory-level or other lower-level observational data. For each stressor the situation is likely to be different. The problem is that neither structure nor function nor composition might serve as an end-point at this level. This means that the type of extrapolations referred to in Table 2.2 may have to be used.

The situation for toxic substances will be developed further by way of example. The problem now is to establish how the common type of laboratory bio-assessment data can be linked to the upper-level phenomena such as structure, function and composition. In laboratory bio-assessments the two most common end-points that can be measured are mortality (*m*) and fertility or fecundity inhibition (*r*) from acute (*a*) and chronic (*c*) toxicity tests respectively.

Inferences [2.7] and [2.8] can be calculated from the conditional probabilities:

$$P(m) = P(m|\bigcup_X aX) \cdot P(\bigcup_X aX) \text{ where X} \in \{\text{T, S, Q, H}\} \qquad [A3.6]$$

$$P(r) = P(r|\bigcup_X cX) \cdot P(\bigcup_X cX) \text{ where X} \in \{\text{T, S, Q, H, N}\} \qquad [A3.7]$$

The last term on the RHS of equations [A3.6] and [A3.7] can then be expanded by using [A3.5]. However, the probability of conjunction (or intersection in set-theoretical terms) in the RHS of [A3.5] can be simplified further if the events *aX* and *cX* are independent.

$$P\left(\bigcup_X aX\right) = \sum_X P(aX) - \sum_{X \neq Y} P(aX \cap aY) + \sum_{X \neq Y \neq Z} P(aX \cap aY \cap aZ) - \ldots \pm P(\bigcap_X aX) \qquad [A3.8]$$

The form for the chronic occurrence of stressors is analogous, with *aX* being replaced by *cX*. If the occurrence of stressors is logically independent, then the intersections are replaced by the product of probabilities (Bain and Engelhardt, 1987).

$$P\left(\bigcup_X aX\right) = \sum_X P(aX) - \sum_{X \neq Y} P(aX)P(aY) + \sum_{X \neq Y \neq Z} P(aX)P(aY)P(aZ) - \ldots \pm \prod_X aX$$

It is known *a priori* that the level and duration of the stressor is dependent on the occurrence of the stressor in the first place. Conventionally, the duration of exposure is assumed to be infinity, i.e. a steady state concentration is assumed. The occurrence of acute stress is assumed to be determined by the level of stressor only. In this case, expressed in set theoretical terms the probability of stress is:

$$\mathrm{P}(aX) = \mathrm{P}(aX \cap alx) = \mathrm{P}(aX \mid alx) \,.\, \mathrm{P}(alx). \qquad [A3.10]$$

Generally though, stressor levels in-stream are variable and consequently the duration of a specific level of stressor is not infinity but of duration τ, where $0 \leq \tau \leq \infty$ or possibly even dynamic. The dynamic case involves mechanistic considerations, which will be considered in Chapter 5. For the purpose of this chapter the level of stressor is assumed to be a function of time but in such a way that τ is long enough for a pseudo steady state to be reached. In analogy to xenobiotics exposure, where it is known that both the level and duration of exposure is important, it is postulated that for all stressors this is true to some extent. Therefore, the expression for the probability if occurrence of stress X due to stressor x should be:

$$P(aX) = P(aX \cap alx \cap adx) = P(aX \mid alx \wedge adx) \,.\, P(alx \cap adx)$$

The level of exposure and duration of exposure are assumed independent. This appears to be reasonable as a first assumption since in general there would be no mechanism that relates the duration and level of exposure. Therefore, the probability of stress becomes:

$$P(aX) = P(aX \mid alx \cap adx) \,.\, P(alx) \,.\, P(adx) \qquad \text{[A3.11]}$$

The problem of determining the risk of unsustainability due to multiple stressors from a single source can be addressed by sequentially solving [A3.10] (or [A3.11] in the case of time-varying concentrations), [A3.9], [A3.6], [A3.7], [A3.4] and [A3.2] (Figure 3.1).

A2.11 NOTES ON THE ESTIMATION OF STRESSOR-RESPONSE RELATIONSHIPS

A2.11.1 TOXIC SUBSTANCE STRESS-RESPONSE RELATIONSHIPS

The aim of this section is to present a method to estimate the parameters for the SRR for toxic substances. In the context used here, toxic substances may refer to any stressor that may be diluted or have its level adjusted when being mixed with water having a different level of stressor. Typically this type of data would be generated by laboratory bio-assessments. Two issues need to be considered: the level of organisation at which the assessment is aimed, the problem of temporally varying stressor levels, and the use of "standard" toxicity benchmarks.

CHOICE OF TEST SPECIES

Not only does the level of organisation within the species of choice matter, but the choice of species also has an influence on the interpretation of derived values. It has become apparent that no single specie can qualify (Kenaga, 1978, Mayer etand Ellersieck, 1986, Blanck *et* al., 1984, Kooijman, 1987). The lowest acute or chronic test result from a set of the most commonly used species, (the alga *Selenastrum capircornutum*, the fish *Poecilia reticulata* and the invertebrate *Daphnia magna*) only managed to come within a factor of 10 of the most sensitive species tested 25% of the time (Sloof *et al.*, 1983; Sloof and Canton, 1983). Therefore, if no single most sensitive species can be found and it is unlikely that a suite of standard test organisms will give an indication of what the susceptibility of the most sensitive species will be like, it could be argued that

a) no species is likely to be significantly more sensitive than the most sensitive test species, or
b) "that differences in sensitivity among species are insignificant unless they are larger than differences among tests of a species-chemical combination" (Suter, 1993), or
c) simply use a safety factor to accommodate all the uncertainty when extrapolating, or
d) assume that species sensitivity will follow some regular distribution and estimate protection levels from that.

The third argument has been in use for some time. The USEPA's uncertainty factor of 10 for taxonomic variance appears to be based on the assumptions that: 1) any invertebrate is a sensitive as *Daphnia* and that any vertebrate is as sensitive as the fish used in the tests, and 2) that protecting a small number of test species 90% of the time is sufficient (Suter, 1993).

The fourth argument recognises the inherent fallacy of the third argument in that there is no evidence that *Daphnia* and fish represent among themselves the most sensitive species or even representative species. The approach used in the derivation of the South African Water Quality Guidelines for the Protection of the Aquatic Environment (Roux, *et al.*, 1996) is based on that used for the calculation of the U.S. National Water Quality Criteria (Stephan *et al.*, 1985) with the exception of the greater emphasis placed on the use of indigenous test species. The approach has been to assume that species sensitivity will follow a regular distribution (in this case a log triangular distribution) and by assuming a level of protection for all species (e.g.95%), a concentration of a toxicant can be calculated. Kooijman (1987) fits a log logistic distribution to toxicity data. However, Suter (1990) considers the choice of distribution to be insignificant in comparison to the more crucial decisions such as level of protection and uncertainties included in the estimation of confidence. It may be argued that all species in a community should be protected and that the selection of any arbitrary protection level does not guarantee protection of ecosystem function. Kooijman (1987) made a similar suggestion. This implies that the criterion value for more and less diverse communities will differ with the more diverse communities having a lower criterion value, since there are more species (and therefore a greater possibility of sensitive species). In contrast, Van Straalen and Denneman (1989) argue that in larger communities the likelihood of functional redundancies is larger and that therefore less restrictive criteria should be applied.

Sensitivity distribution based on species distribution assumes that test species are randomly drawn from the community they are supposed to represent. The argument has been raised that clearly test organisms are not randomly selected, but are usually selected on the basis of ease of laboratory cultivation and happenstance (Cairns and Pratt, 1989). However, ease of laboratory cultivation is determined by species specific knowledge and good laboratory technique rather than by species sensitivity as is borne out by the observation that sensitive species survive and thrive under natural conditions which are considerably more adverse than laboratory conditions. Therefore, unless specifically contraindicated, there would be sufficient reason to assume random selection of species in the toxicity test data to warrant using the data to estimate the probability density function parameters.

Estimating parameters for distributions normally requires a considerable amount of data, which is often lacking. There is considerable need to use extrapolation to derive parameters in sparse data sets. If there are too few data to confidently estimate the parameters of the distribution (such as NOEC, EC_{50} and another percentile < 50) of sensitivity of species for a chemical, it can be estimated by considering the sensitivity data across chemicals where the relevant data are available.

INDIVIDUAL VS. POPULATION BASED ASSESSMENT

The individual based approach in ecology is essentially an application of the reductionist methodology. There are two approaches to follow in conceptualising populations:

1) The **population approach** where the whole population consists of individual organisms that are essentially identical subject to natality and death. An example is the common Lotka-Volterra models used with some success in explaining at a phenomenological level the changes in predator and prey fish caught after the first world war (Braun, 1983 pp 441-449; Suter, 1993). This type of model does not necessarily demonstrate the dynamics involved at a *biologically measurable* level. The parameters in these models (e.g. the predation rate, competition intensity etc.) are mathematical descriptors that are not directly measurable, but can only be inferred or calculated from real population measurements.
2) **Individual based models,** where it is recognised that a population may consist of a number of individuals with different ages, morphological characteristics, fecundity, mortality rates etc. The individual based methods in population ecology explicitly incorporates a knowledge of dynamics and socio-biology of populations in terms of biologically significant parameters such as fecundity, mortality rates or survival probability (Lomnicki, 1992).

A stressor will generally affect different life stages of an organism in different ways, and the effect on the population as a whole can usually not be assessed from "standard" toxicity benchmarks such as the LC50 (Lenski and Service, 1982; Mayer, et al., 1989; Caswell, 1996). The individual-based bio-assessments depend on the testing of a cohort of organisms usually for a relatively small fraction of their natural lifetime. Even chronic toxicity tests do not combine mortality and fecundity data to estimate impacts on a population. In order to do this, though; the life history of the organisms as well as the survival and fecundity rates of a cohort of the organism needs to be known.

A well-established approach to estimate population level effects from individual level observations is by using demographic population models (Caswell and John, 1992). Knowledge of the individual state (i-state) variables such as age size and physiological state are used to derive the population state (p-state). Construction of a population model requires a function that combines the current p-state dynamics and the environment The types of models that could be involved are described in Table A2.1. The discrete-state, discrete-time model described by Caswell (1989) was chosen because the type of data generated in a laboratory bio-assessment appears to fit this model better than the continuous time models.

Table A2.1. Mathematical frameworks for p-state variable models

p-State	Time	Model Type	Reference
Discrete	Discrete	Projection matrices	Caswell, 1989
Discrete	Continuous	Delay-differential equations	Nisbett and Gurney, 1982
Continuous	Continuous	Partial differential equation	Metz and Diekman, 1986

Where individuals can be differentiated on some basis or another, the population projection matrix model Eq. [A2.11] gives the conditional expectation of population number per class (expressed as the vector **n**(t)):

$$E(\boldsymbol{n}(t+1)|\boldsymbol{n}(t)) = \boldsymbol{A}\cdot\boldsymbol{n}(t) \qquad \text{[A2.11]}$$

An inherent advantage in this type of model is the underlying stochastic description of a population already incorporated in the model. From Eq. [A2.11] the assumption of Markov-chain conditions is apparent. This may be a drawback since the future state of a population is not always only dependent on its present state, but may be dependent to some extent on its recent history. As a first approximation the Markov condition may be sufficient. The model can be formulated by a matrix equation Eq. [A2.12].

$$\mathbf{N}_t = \mathbf{A}^t \cdot \mathbf{N}_0 \qquad \text{[A2.12]}$$

where: $\mathbf{A} = \begin{bmatrix} F_1 & F_2 & \cdots & F_s \\ P_1 & 0 & \cdots & 0 \\ \vdots & \vdots & \vdots & \vdots \\ 0 & 0 & P_{s-1} & 0 \end{bmatrix}$

and P_i is the probability of survival of members of age class i. Fertility of the population is described in terms of fertility coefficients F_i

A population that responds according to this model will (Caswell, 1989):

1. Eventually reach a **stable age distribution**
2. Grow or decline at a **constant rate**, and
3. Have its long-term behaviour determined by its **dominant eigen value**.

The utility of the transition matrix $\boldsymbol{A}$ in ecotoxicology lies in:

(a) The connection between the dominant eigenvalue of $\boldsymbol{A}$ and the intrinsic rate of population growth. If λ_1 is the dominant eigenvalue of the transition matrix $\boldsymbol{A}$, then $\lambda_1 = e^r$ with r the nominal rate of population growth (Caswell, 1989).

(b) The p-state parameters are inferred from easily measured i-state transition variables. In the case of aquatic toxicity tests these are measured in the form of fecundity and survival rates or probabilities.

The SRR parameters can be estimated from an assessment of the population growth characteristics projected from the survival and fertility data collected from individual organisms.

The **upper acceptability limit** (the catastrophic effect level) can be said to be the minimum stressor level corresponding to a zero population growth rate. The rationale for this is that if population numbers are expected to decline in the absence of natural processes such as competition and predation, then the effect could only be expected to be worse in the presence of such factors.

The **lower acceptability limit** (no-observable-effect level) is not as easily assessed since there is no natural cut-off point. In order to generate such a cut-off point it would be necessary to make some value judgements. It could, for example, be argued that any observable decline in population growth rate r would be unacceptable. This r would be the growth rate that could be resolved from the natural population growth rate r_0 with a confidence of, say, 90% (α = 0.1). This rationale is similar to that used in the definition of a toxicity NOEC, subject to the same type criticism, i.e. that statistical significance has nothing to do with ecological significance (Suter, 1993). This argument is valid if there is sufficient ecological knowledge available to estimate an ecologically significant value of r. If not, the statistical value must act as surrogate for ecological significance.

Eq [A2.12] represents a general population growth assessment. In order to use this type of model, there are two types of parameters that need to be calculated or estimated: the age-specific **probability of survival** and the age-specific **fertility functions**.

Survival probability estimate

One of the most powerful means to generate these data is by using hazard analysis (Cox and Oakes, 1984). A hazard model relates the probability of a transition occurring (as the dependent variable) to a causal factor (as the independent variable). If $f(t)$ is the instantaneous probability of an event occurring at time t and $F(t)$ is the cumulative probability of the event having occurred before time t, then the hazard function, $\mu(t)$, for example the probability of an organism dying in between t and $t+dt$ is given by (Caswell, 1989), is given by Eq. [A2.13].

$$\mu(t)dt \approx -dt\frac{\partial \ln l(t)}{\partial t} \qquad \text{[A2.13]}$$

where $l(t)$ is the probability of surviving to time t. Generally, the probability of surviving to time t give exposure to concentration x, $S(t/x)$, is related to the hazard function $h(t/x)$ by (Namboodiri and Suchindaran, 1987; Moore, *et al.*, 1990):

$S(t/x) = \exp[-\int_0^t h(t/x)\partial t]$. The hazard function $h(t|x)$ is also called the force of mortality and is equivalent to $\mu(t)$ used by Caswell (*op cit.*). From Eq.[A2.13] the probability of survival over the interval $t+\Delta t$ is given by Eq. [A2.14].

$$\frac{l(t+\Delta t)}{l(t)} = e^{-\mu(t)\Delta t} \qquad \text{[A2.14]}$$

Using a proportional hazards model, the fraction (probability) survival under a given exposure regime $S_1(t)$ can be related to the baseline survival $S_0(t)$ by (Namboodiri and Suchindaran, 1987): $S_1(t) = S_0(t)^{\exp[X\beta]}$.

In order to parameterise the population transition matrix $\boldsymbol{A}$ of Eq. [A2.12] it is necessary to estimate $S_1(t_i)$ for each time interval t_i and each life stage modelled in this matrix. There are two options to estimate the survival:

a) by **direct calculation** from suitable **experimental data** (e.g. from toxicity bio-assessment) where $S_1(t_i)$ and $S_0(t_i)$ can be calculated from the exposed and control runs respectively, or
b) by **indirect estimation** when no suitable life table experimental data are available where $S_1(t_i)$ must be calculated from **other ecotoxicological data**.

Direct calculation from bio-assessment data

By curve fitting the parameters for the proportional hazards model could be determined. Moore, *et al.* (1990) tested a model of the form $P_i(x) = P_{0,i}^{\exp[\beta_i(x-x_0)]}$ and showed that for three tested pesticides the potency β remained constant through all intervals, and hence β_i can be replaced by

β. Here $S_i(x)$ is the probability that a test animal alive at the beginning of the i^{th} interval will survive to the end of the i^{th} interval, β_i is the potency of concentration x during the i^{th} interval, x_0 is an arbitrarily chosen log concentration to centre the observations and P_i is the underlying conditional probability of survival at the centring concentration x_0.

$$S_i(x) = \left(\prod_{j=1}^{i} P_j(x) \right)^{\exp[\beta(x-x_0)]} \qquad \text{[A2.15]}$$

Bio-assessment data that would be applicable for this kind of estimate would result from experiments where a suitable life table can be generated. This would mean that:

- the exposure would encompass practically the whole life cycle of the organism, or at least that part of the life cycle spent in water, and
- both mortality and fertility data need to be recorded, which means that range of exposure levels need to be wide enough.

Indirect estimation from other ecotoxicological data

The survival can be estimated from fundamental ecotoxicological data such as the uptake and excretion rates, the lethal body burden and the log K_{ow} of the substances involved. The methodology is similar to survival time analysis. The toxicokinetics become important when estimating the fraction of a population surviving to a given time. The time would typically correspond to the cohort age structure used to discretise the lifetime of the organism. The calculation uses the same type of data used to estimate the effect of temporally varying concentrations.

THE PROBLEM OF TEMPORALLY VARYING COMPOSITION

In the derivation of substance specific criteria bio-assessment data was used that selected the standard test durations (e.g. 48 hours for many of the smaller invertebrates and 96 hours for larger animals). In these tests the levels of substances were kept constant. Stressor levels cannot be expected to be constant in real situations. This begs the question of what happens when stressor levels vary. The approach in the application of the USA criteria has been to use 1-hour average concentrations when considering acute substance specific criteria and to use 4-day average concentrations when using chronic criteria (Delos, 1994).

In order to clarify the role of time in the effect assessment of substances, the toxicokinetics need to be considered. This involves determining the mode of action (MOA). Depending on the classification used anything between two and eight MOA's can be distinguished (Verhaar, *et al.*, 1999). These may include the narcotics, polar narcotics, electrophiles and reactive or receptor mediated compounds. Among non-metal toxicants the polar narcotics probably represent the most rapidly excreted substances and the reactive chemicals the least excreted compounds. Mechanistically these classes are distinct and a comparison appears in Table A2.2

Table A2.2 Comparison of polar narcosis and reactive toxicity (Legierse et al., *1999; Freidig,* et al., *1999; Verhaar,* et al., *1999)*

Aspect	Polar Narcosis	Reactive toxicity
Receptor interaction:	Reversible	Irreversible
Toxicodynamics determined by:	Cell membrane	Intracellular chemical pool
Dose metric	Internal concentration	Area under concentration vs. time curve (AUC)
Critical physiological parameter	Critical body residue (CBR) or lethal body burden (LBB) = constant for all chemicals in class	Critical area under curve (CAUC) = constant (CBR is temporally variable)
$EC_{50}(t)$ determined by:	Bioconcentration kinetics	Cumulative inhibition of receptor
Model	CBR	Critical Target Occupation (CTO)
$LC_{50}(t)$ =	$\dfrac{LBB}{BCF \cdot (1-e^{-k_2 t})} = \dfrac{LC_{50}(\infty)}{(1-e^{-k_2 t})}$	$\dfrac{CUAC_a}{t} + LC_{50}(\infty)$
LBB =	$LC_{50}(\infty) \cdot BCF$	$BCF \cdot (1-e^{-k_2 t}) \cdot LC_{50}(t)$

With complex effluents, variables such as the LBB cannot be determined unless the effluent composition is known; an exercise that would partially defeat the purpose of using WET assessment in the first place. However, from the expressions in Table A2.2, there is a relationship between the LBB and the $LC_{50}(\infty)$. For the purpose of evaluating the age-specific cumulative fractional mortality it is necessary to know $LC_S(t)$.

Mancini (1983) developed a simple toxicokinetic approach to estimate effect for time varying concentrations. Based on the assumptions that:

1. Variation in survival times defines a distribution of sensitivity
2. At any concentration the same percentile survival time defines a common sensitivity level, and
3. All organisms with similar sensitivity have similar regulatory characteristics

and using a simple single compartment model for the target organism:

$$\frac{dy(t)}{dt} = k_1 \cdot x - k_2 \cdot y(t) \qquad \text{[A2.16]}$$

where y = intra-organism concentration of the toxicant [mass toxicant/mass organism]

x = concentration of toxicant in the water [mass toxicant/volume water]

k_1 = uptake rate [volume water/(mass organism*time)]

k_2 = depuration rate [/time]

with the boundary values:

$y(0) = 0$

$y(t') = d$ (lethal dose)

where: t′ = time to death. If at first it is assumed that the concentration is constant for a period it was shown that:

$$y(t) = \frac{k_1}{k_2} \cdot x \cdot \left[1 - e^{-k_2 t}\right] \qquad \text{[A2.17]}$$

Recognising that $BCF = \frac{k_1}{k_2}$ Eq [A2.17] rearranges to Eq. [2.18].

$$y(t) = BCF \cdot x \cdot \left[1 - e^{-k_2 t}\right] \qquad \text{[A2.18]}$$

When the intrabody dose, $y(t)$, reaches a level referred to as the critical body residue (CBR) or lethal body burden (LBB), the organism dies. The implication is that different chemicals with a narcotic mode of action will display an additive body burden, which, on reaching the LBB for the organism, will result in death of the organism. (Sijm *et al.*, 1993). For anaesthetic chemicals the LBB appears to vary between 2 and 8 mmol/kg irrespective of structure.

SOME EXPRESSIONS FOR THE BODY RESIDUE OF NARCOTIC SUBSTANCES UNDER TEMPORALLY VARIABLE WATER CONCENTRATIONS

For **pulsed toxicant concentration with a square waveform**: with water concentration x for $0<t<t_0$
and x=0 for $t_0<t<t_1$. Then:

$$C(t_1) = \frac{y(t_1)}{u} = \frac{x}{r} \cdot \left[e^{-r(t_1-t_0)} - e^{-rt_1}\right] \qquad \text{[A2.19a]}$$

or (Mancini, 1983)

$$C(t_1) = \frac{y(t_1)}{u} = \frac{x}{r} \cdot \left[1 - e^{-rt_1}\right] + C(t_0) \cdot e^{-rt_1} \qquad \text{[A2.19b]}$$

This corresponds to a situation where depuration takes place when the external concentration drops after uptake of toxicant at the higher ambient toxicant concentration (Figure A2.1). If toxicant build-up takes place long enough, then that fraction of organisms for which the equivalent dose, $C(t)$, equals or exceeds the equivalent mortality dose, D, die.

This could have a significant effect on the mortality of the organism. Considering Figure A2.1, if the equivalent mortality dose is 10 mmol/kg, then the expected survival time for the 10 percentile of organisms is about 14 days, the median survival time is about16 days, but the 90th percentile of organisms in this exposure scenario is ∞.

More generally, if the aqueous concentration varies in a stepwise manner with changes at discrete time points t_i a fixed time interval t_d apart and with the concentration remaining constant during this period at x_i, then the internal concentration at the end of the interval is given by (Kooijmans, 1994):

$$y_{t+1} = e^{-rt_d} \cdot y_t + (1 - e^{-rt_d}) \cdot \frac{x_t \cdot u}{r} \qquad \text{[A2.20]}$$

If x_t follows a random increment process, then solution of the stochastic analogue of the differential equation [5.10] yields the expected value of $y(t+1)$ is:

$$E[y_{t+1}] = (e^{-rt_d})^{t+1} \cdot E[y(0)] + (1 - e^{-rt_d}) \cdot \frac{u}{r} \cdot E[x_t] \cdot \sum_{j=0}^{t} (e^{-rt_d})^j \qquad \text{[A2.21]}$$

and

$$\mathrm{var}[y_t] = \mathrm{var}[x_t] \cdot \left(\frac{u}{r}\right)^2 \cdot \frac{1 - e^{-rt_d}}{1 + e^{-rt_d}} \qquad \text{[A2.22]}$$

In continuous time the expected value of $y(t)$, $E[y(t)]$, is the same as $E[y_t]$ in equation [A2.21] and:

$$\mathrm{var}[y(t)] = \mathrm{var}[x_t]\left(\frac{u}{r}\right)^2 \left(1 - \frac{1 - e^{-rt_d}}{rt_d}\right) \qquad \text{[A2.23]}$$

For water **concentrations with an exponential decay** function (peak concentration A and decay constant k), i.e. $x(t) = A.e^{-kt}$, :

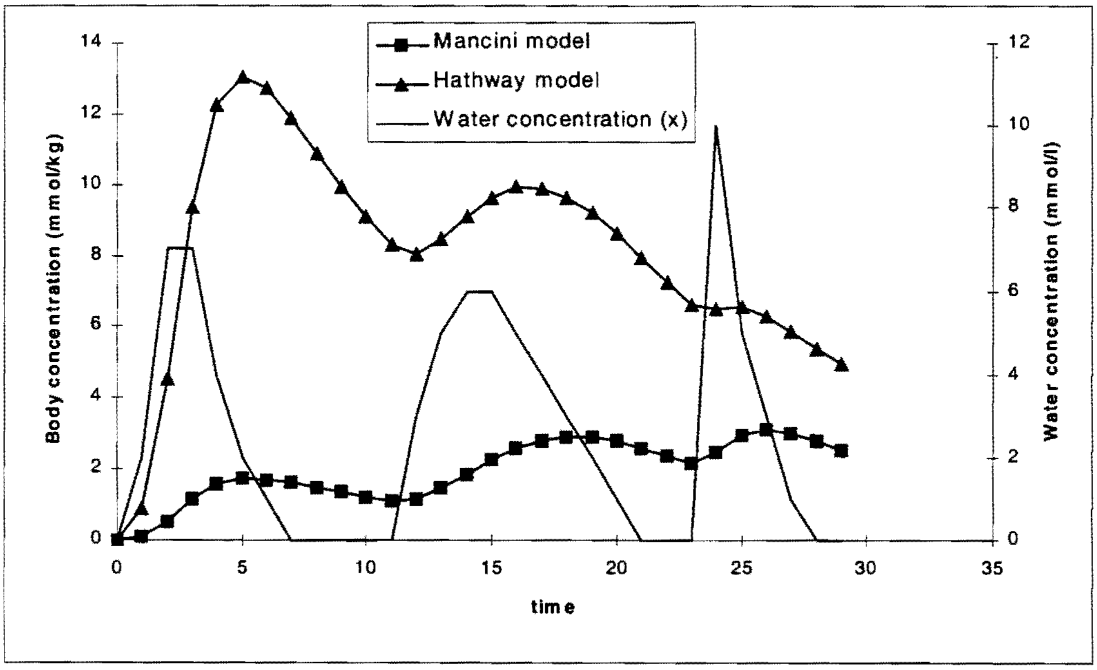


Figure A2.1 An illustration of the importance of knowledge of mechanisms of toxicology. The body burden of a hypothetical substance with k2=0.09 and BCF = 1.11 predicted by the Mancini and Hathway models (Eqs. [A2.19a] and [A2.26] respectively) as a function of the substance concentration in water. The Mancini model predicts a more rapid response to changes in aqueous concentration.

$$D = \frac{d}{u} = \frac{A}{(r-k)} \cdot \left[e^{-kt'} - e^{-rt'}\right] \qquad \text{[A2.24]}$$

and

$$C = \frac{y(t)}{u} = \frac{A}{(r-k)} \cdot \left[e^{-kt} - e^{-rt}\right] \qquad \text{[A2.25]}$$

The uptake of a substance has so far been assumed to be instantaneous. This would generally not be true and Hathway (1984) suggested that the equilibrium concentration may be described by:

$$\frac{dy}{dt} = x \cdot u \cdot e^{-ut} - ry \qquad \text{[A2.26]}$$

The effect of this model is that the organism does not immediately respond to a change in concentration. If the dosed concentration, x, is a function of time, x(t), then a lagging of intra-organismal concentration of the toxic substance can be expected. This is demonstrated in Figure A2.2.

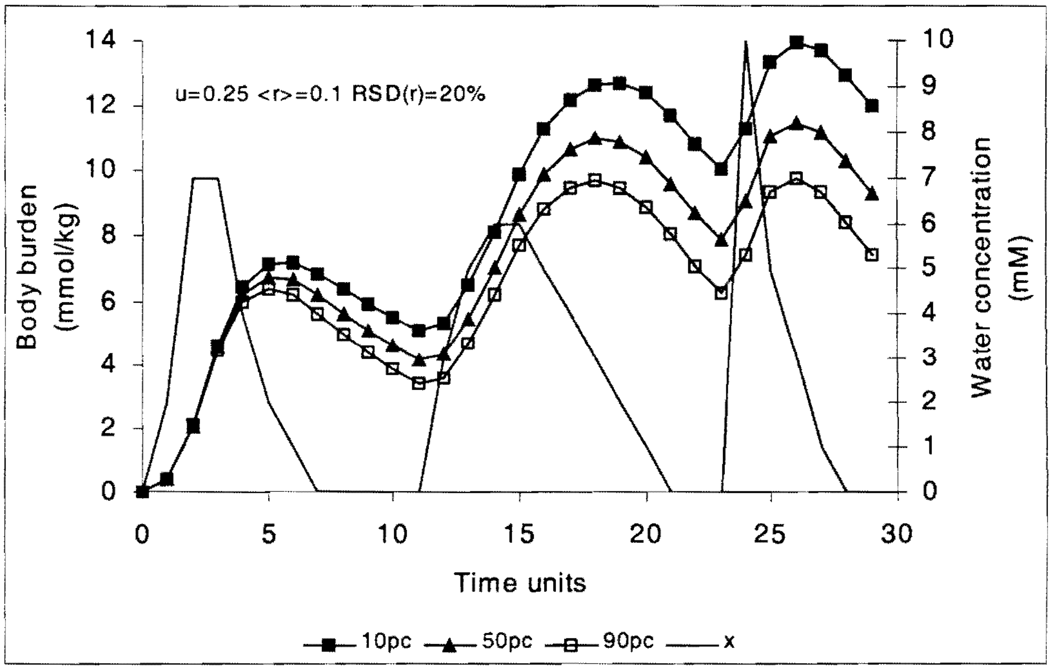


Figure A2.2 Demonstrating the effect of variability in individual organism depuration rate on the expected body burden within for a population as a function of concentration in water. Body residue of a hypothetical substance in an organism with an average k2 = 0.1 and standard deviation = 0.02. The average BCF = 4.

A further refinement can be attained by recognizing that the substance(s) absorbed may not in themselves be toxic and that further reaction inside the organism, whether by activation or binding to a target receptor, may be required to see an effect. For a reaction between dosed substance A and intra-organismal substance B:

$$A + B \underset{k_3}{\overset{k_2}{\rightleftharpoons}} AB \qquad \text{[A2.27a]}$$

Then, with respect to A, at a nominal concentration y_1, the concentration of the reaction product (AB) y_2 is given by:

$$\frac{dy_2}{dt} = k_2(y_1 - y_2) - k_3 y_2 \qquad \text{[A2.27b]}$$

If the kinetics is determined by the concentration of y_1, i.e. the uptake of the toxicant is rate determining, then the effect will be determined by Eq. [A2.27c]. If the concentration of the receptor, b, is rate determining then the effect will be determined by [A2.27d]

$$y_2 = \frac{k_2 y_1}{k_2 + k_3} \cdot \left[1 - e^{-(k_2+k_3)t}\right] \qquad \text{[A2.27c]}$$

$$y_2 = \frac{k_2 b}{k_2 + k_3} \cdot \left[1 - e^{-(k_2+k_3)t}\right] \qquad \text{[A2.27d]}$$

The dynamics of toxic effect of substance A applied *in aquo* at concentration x is give by the system of equations [A2.28].

$$\begin{aligned} \frac{dy_1}{dt} &= u \cdot f \circ x(t) - r \cdot y_1(t) \\ \frac{dy_2}{dt} &= k_2(y_1 - y_2) - k_3 y_2 \end{aligned} \qquad \text{[A2.28]}$$

with the driving function $f(x)$ taking on a suitable form.

Alternative to Mancini's assumption that there is a distribution of regulatory efficiency that gives rise to variability in response, it could be argued that regulatory efficiency is constant but that there is distribution of receptor site density over a population, i.e.. that b in Eq. [A2.27d] is stochastic variable.

FURTHER RESEARCH

In the case of single substances, the above approach is simple to quantify in principle since the body burden of an identifiable substance can be measured and k_2 and BCF can be calculated. The problem arises in predicting the effect of temporally varying complex effluents. As shown in the foregoing illustration the body burden of a substance in an organism varies with varying ambient concentration.

The problem that needs to be solved is how to estimate the body burden of lethal components of a complex mixture from toxicity bio-assessments. If it is assumed that the components of a mixture interacts by the narcotic mechanism, then at the time of death of an organism, the narcotic substances that had partitioned from the mixture and of which the organism cannot excrete fast enough, will total to the LBB. It seems reasonable to suppose that a complex effluent will have an apparent k_2 value. If this value is known, then Eq. [A2.25] (or A2.19 to A2.26 above depending on the situation) can be used to estimate the apparent body burden of the mixture (effluent). If it is recalled that k_2 is a stochastic variable for a population, then the probability distribution of mortality can be estimated and from that the organisms population growth can be estimates (subject to assumptions or measurements about it fertility). The two critical questions that need to be answered are:

- How can the apparent BCF of a complex mixture be estimated?

- Can the differential excretion rate for the components be estimated from measurement other than by temporally variable toxicity-bioassessment?

In both cases the development work on biomimetic extractions seems encouraging (Verbruggen, *et al.*, 1999) and could be investigated further.

A2.11.2 HABITAT- AND FLOW-STRESSOR-RESPONSE RELATIONSHIPS

The prediction of biological effect is notoriously difficult and yet the need for prediction is very real (Armitage, 1994). The problem of flow and habitat stress assessment has been presented in Chapter 3 as a strong reason for the use of fuzzy set theory. The reason being that often there is no controlled experimental evidence to derive the SRR parameters. These parameters are estimated based on the assessment of an expert based on analogy, limited observation etc. The situation is analogous to what is described by Klir and Folger (1988) as an interpersonal communication problem. The stressor risk assessment can be formulated in the form $E = R \circ A$ where R is the fuzzy relationship between fuzzy stressor situation analysis A and the fuzzy expectation of effect E and o is a suitable implication operator. In the examples presented in this study, R has been simplified to crisp relationship but this need not generally be so.

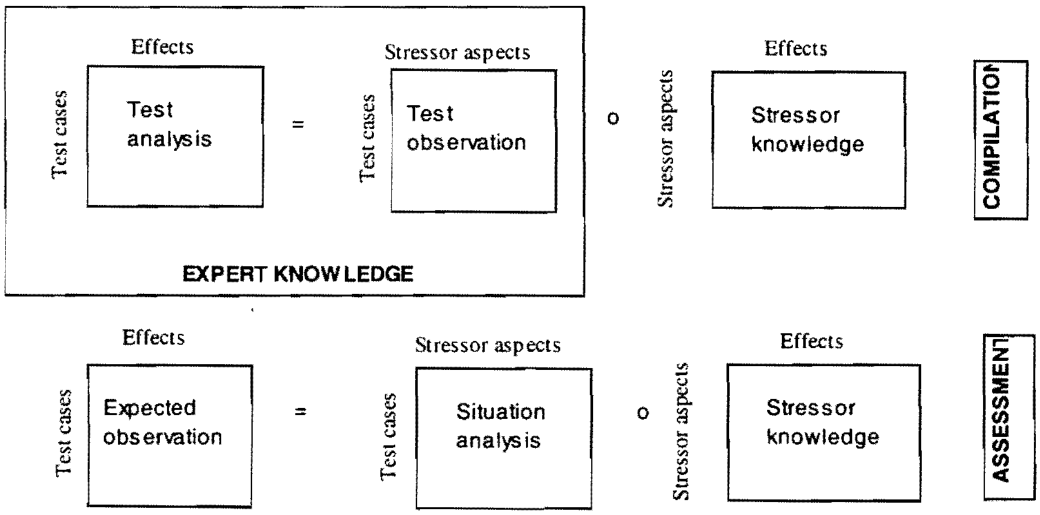


Figure A2.3. Schematic of the possible use of fuzzy sets in assessing fuzzy expectation. Stressor-effect relationships are encapsulated in the stressor knowledge block. The upper schematic follows a logic from left to right (i.e. the stressor knowledge is generated) while in the lower schematic the goal is deriving the expectation of effect for a give number of cases.

The expectation assessment problem resolves into two practical problems: 1) Deriving the relationship R expressing the knowledge of response of stressors, and 2) incorporating new ob servational evidence to update the expectation E.

NOTES ON THE FUZZY RELATIONSHIP R

R is derived from a training set of stimuli and responses collected over as wide a range as possible of test cases. The process is described in Figure A2.3

Both R and X are derived from and informed by the interpretation of real data by the ecologist/ ecotoxicologist. Consider the situation where the stressor is characterised by characteristic set $X = \{x_1, x_2, \ldots x_n\}$ while the effects are characterised by set $Y = \{y_1, y_2, \ldots y_m\}$. The experience of the ecologist in dealing with a particular stressor derives from observations in a number of test cases with corresponding stressor situation analyses. For every stressor metric $x \in X$ there is an observed or inferred response $y \in Y$ in the set of test situations $T = \{t_1, t_2, \ldots t_k\}$. Each test case t results in a stressor knowledge matrix:

$$R_t = \begin{bmatrix} \mu_R(x_1, y_1) & \cdots & \mu_R(x_1, y_m) \\ \vdots & \mu_R(x_i, y_j) & \vdots \\ \mu_R(x_n, y_1) & \cdots & \mu_R(x_n, y_m) \end{bmatrix}$$

The elements of the knowledge matrix R can be evaluated in two different ways resulting in two different knowledge bases:

- An occurrence relation R_o that corresponds to the answer to the question: "How often does stressor characteristic x occur in conjunction with effect y?" This is derived from an assessment over all the test cases of the frequency of the co-occurrence of x and y, or,
- An confirmatory relation R_c that corresponds to the answer to the question: "How strongly does effect y confirm the presence of stressor characteristic x?". This results from an analysis of the correlation of the intensity of x and the intensity of y.

This approach could be considerably expanded, both in terms of the information content of the knowledge base and the modelling of, and expert query to construct the relationship (see e.g. Yager (1992)).

EXAMPLE: FLOW-RELATED EFFECT ASSESSMENT

Consider a flow-related stressor characteristic set $= \{$*sufficient water depth (d), correct flow timing (t), adequate scour flow(s)*$\}$ and the effect characteristic set $= \{$*adequate fish community maintenance (f), adequate invertebrate community maintenance (c), physical stream habitat maintenance (h), refugia maintenance (r)*$\}$. The linguistic qualifiers and their membership interpretation are listed in Table A2.2.

If the modifier "very" needs to be added to the qualifier then the modified membership function, $\mu_A'(x) = 1.5{*}\mu_A(x)-0.25$. Assume a knowledge base in form :

d and t are very often important for f ,

f is always important for c,

s is seldom important for f,

s is often important for c,

s is always important for h,

d is sometimes important for h,

t is seldom important for c,

t is never important for h, etc.

Table A2.2 Linguistic qualifiers and their membership grade evaluation

Characteristic (x or y) qualifier	$\mu_A(x)$
Never	0
Seldom	0.25
Sometimes	0.5
Often	0.75
Always	1

From these data a relationship can be constructed:

$$R = \begin{array}{c|cccc} & f & c & h & r \\ d & 0.75 & 0.75 & 0.5 & 0.25 \\ t & 0.875 & 0.25 & 0.75 & 0.125 \\ s & 0.125 & 0.875 & 1 & 0.125 \end{array}$$

When a specific flow scenario is being assessed, the probability distribution for the flow characteristics might be assessed from the knowledge of the catchment size and topography, rainfall record or from actual measurements. The values of $\mu_A(x)$ will likely be derived from an expert assessment of when the measured or predicted flow corresponds to sufficient depth, suitable timing and adequate scour flow. A typical example of this type of expert knowledge encapsulation might be as shown in Table A2.3. This implies that a relationship exists that expresses $\mu_A(x)$ as a function of flow. A typical flow assessment A might be:

$$A = \begin{array}{ccc} d & t & s \\ 0.5 & 0.7 & 0.2 \end{array}$$ if d = 30 cm, t = 7.6 weeks and s = 6.8 m^3s^{-1}.

If the max-min composition is used as implication operator then the expected effect will be:

$$E = \begin{array}{cccc} f & c & h & r \\ 0.7 & 0.5 & 0.7 & 0.25 \end{array},$$ which means that refugia maintenance is most likely to be affected by the affected flow scenario. If all effects are assumed to be equally important in determining the end-point effect (e.g. loss of sustainability), the possibility for the end-point will be (1-min($\mu_A(x)$)) = 0.75.

Table A2.3. Example of a possible format of membership functions for flow stressor characteristics.

Characteristic	Metric	Function
Depth (*d*)	Average flow depth (cm)	$\mu_A(d)=\begin{cases}1 & if\ d>50\\ \dfrac{50-d}{40} & if\ 10\le d\le 50\\ 0 & if\ d<10\end{cases}$
Timing (*t*)	Displacement of expected peak flow (weeks)	$\mu_A(t)=\begin{cases}1 & if\ t<2\\ \dfrac{t-2}{8} & if\ 2\le t\le 10\\ 0 & if\ t>10\end{cases}$
Scour flow (*s*)	Minimum flow rate ($m^3.s^{-1}$)	$\mu_A(s)=\begin{cases}1 & if\ d>0.8\\ \dfrac{0.8-s}{0.6} & if\ 0.2\le s\le 0.8\\ 0 & if\ s<0.8\end{cases}$

EXAMPLE: ESTIMATING ACCEPTABLE STRESSOR VALUES.

The same data as in the previous example applies. In order to derive management criteria, the process for the assessment above is reversed in that an acceptable level of effect is specified while the corresponding stressor level is required. Say that a level α of effect is considered acceptable. That means that $\mu_E(y) = \alpha$, which implies that $\alpha = \max_{x\in X}[\min(\mu_A(x), \mu_R(x,y)]$.

This means that $\min_{x\in X, y\in Y}\{\mu_A(x), \mu_R(x,y)\} \le \alpha$ or,

$$\mu_A(x) \le \begin{cases}\alpha & if\ \mu_R(x,y)\ge\alpha\\ \mu_R(x,y) & if\ \mu_R(x,y)<\alpha\end{cases} \qquad [A2.29]$$

Therefore, if $\alpha = 0.2$ then $\mu_A(d) \le 0.2$, $\mu_A(t) \le 0.125$ and $\mu_A(s) \le 0.125$, which translates to d = 42 cm, t = 3 weeks and s = 0.725 $m^3.s^{-1}$.

A11.3 INTEGRATING BIOMONITORING IN ECOLOGICAL EFFECT EXPECTATION

The previous section had shown that the estimate of $\mu_A(x)$ is very important in both effect assessment and stressor value assessment. The function parameters illustrated in Table A2.3 will determine to large extent what the outcome a calculation will be. At the outset, before any site-specific data are available, these parameter values stem from analogy or even educated guessing. In either case there is room for uncertainty in the parameters.

For flow-related or habitat stress, it is unlikely that experimental values will (generally) be available. However, a number of biotic indices have been developed that pronounce on the stressor impacts to greater or lesser extent (Metcalfe-Smith, 1994, Kleynhans, 1999a). These data are often the only indication of *in situ* effect that is available for estimating SRR's. These biomonitoring data may be therefore be useful in informing and updating effect.

This situation may be modelled as being analogous to the combination of evidence from evidence theory. An application of Dempster's rule of combination (Eq. [A2.30]) as described in Klir and Folger (1988) will be used to illustrate how biomonitoring results can be used to update SRR parameters (see also Smets, 1991a, b and c).

$$\mu_{12}(A) = \frac{\sum_{B \cap C = A} \mu_1(B) \cdot \mu_2(C)}{1 - \sum_{B \cap C = \varnothing} \mu_1(B) \cdot \mu_2(C)} \qquad \text{[A2.30]}$$

where two independent sets of evidence (or expert opinion) on sets A, B and C.

Consider the case where there are fish community integrity (*fi*) data and invertebrate community integrity (*ii*) available and instream habitat integrity (*hi*) data. These data may be interpreted by an expert as indicating that the SRR must be adjusted (set D) to indicate lower effect (L), higher effect (H) or no substantial change (N). The combined evidence can be used to generate a membership function for each set as indicated in Table A2.4 below.

Table A2.4 Evaluating the membership from biomonitoring data.

Biomonitoring qualitative indication	Change assessment	Membership
↑↑↑ or ↓↓↓	Definitely	1
↑↑ - or ↓↓ -	Likely	0.75
↑ - - or ↓ - -	Maybe	0.5
↑↓ -	Unlikely	0.25
- - -	No	0

↑, ↓ and – indicate evidence upward, downward and no adjustment respectively.

If the modifier "very" needs to be added to the qualifier then the modified membership function, $\mu_A'(x) = 1.5*\mu_A(x)-0.25$. For the purpose of this evaluation it is assumed that L ∪ H (lower or higher) and L ∪ N ∪ H (lower or higher on no change) are empty sets.

It is now assumed that the current parameter set is the accepted set since no *a priori* evidence exists that this set should be changed in any particular way. This is interpreted to mean that the evidence is equally distributed over all the changes that need to be made and therefore $m_I(D)$ =

0.2 (i.e. the evidence is equally distributed over the 5 cases in Table A2.4). The other evidence for change ($m_2(D)$) is derived from the biomonitoring data membership $\mu_D(x)$ (Table A2.4). In order to meet the requirement for evidence that

$$\sum m(X) = 1, \qquad m(X) = \frac{\mu_D(x)}{\sum \mu_D(x)}$$

An example of an update is provided in Table A2.5.

Table A2.5 An example of evaluating evidence for the change of SRR parameters.

Change	m_1	μ_D	m_2	m_{12}
L	0.2	0.75	0.4	0.45
H	0.2	0.125	0.07	0.08
N	0.2	0.5	0.27	0.18
L ∪ N	0.2	0.25	0.13	0.28
H ∪ N	0.2	0.25	0.13	0.01

The implication of the values in Table 2.5 is that SRR parameters are most likely to be adjusted for lower response but they might also stay the same. As a first (unsophisticated) approach parameter values in Table A2.3 might be iteratively adjusted until m_{12} in table A2.5 indicates neutrality with respect to the need for adjustment.

The indications are that the Dempster-Schafer approach can be used to update the SRR's of flow and habitat related stressors from biomonitoring results. The details of these procedures need to investigated.

APPENDIX TO CHAPTER 4

NOTES ON THE SOLUTION TO THE DSMS PROBLEM

A4.1 CODING OPTIONS IN THE SOLUTION OF THE DSMS PROBLEM BY GENETIC ALGORITHM OPTIMISATION

The formulation of the optimisation problem is described in Paper 4. The problem was coded in MS-DOS QBasic (Version 1.1). This choice of coding language was solely dictated by familiarity and not by any considerations of efficiency of programming. The coding for the various versions of the algorithms is listed in the Addendum.

Four versions of the genetic algorithm coding were produced. The approaches and their differences are described in Table A4.1.

Table A4.1 Differences in versions of the genetic algorithm for the solution of the catchment optimisation problem investigated in this study. Coding name refers to listing in the Appendix of this chapter.

Coding name	Attenuation satisfaction (λ_x)	Equity constraint used?	Control parameter initialisation distribution
G1A	Average {source minima}	No	Uniform from focussed or shifted parameter domain
G1B	Average {source minima}	No	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.

G2B	Average {source minima}	Yes	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.
G3B	Inf{source minima}	Yes	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.
G4B	Inf{source minima}	No	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.

A4.2 RESULTS

APPLICATION OF A GENETIC ALGORITHM TO THE CATCHMENT DSMS PROBLEM

The results of algorithm convergence and the control variables corresponding to the best λ value are shown in Figures A4.1 to A4.7.

Comparison of Figures A4.1 a) and b) indicates that there are probably two minima with λ values 0.54 and 0.74 with the latter probably representing the optimum. There is a slight improvement in the rate of convergence of the algorithm using all exponential distributions to assign initialising values to control variables. The probability of finding the optimum is slightly lower in the former. Comparison of the optimal attenuation values indicates similar performance. The slightly better convergence rate favoured using the exponential distribution in further work. Comparison of λ with λ_X and λ_R (not showed here) indicated that λ_X was the dominant factor in determining λ.

The argument might be made that optimisation with the constraints as given treats different sources of the same stressor differently. Including the equity constraint produced results as shown in Figure A4.2. The addition of the equity constraint significantly reduced the rate of convergence (Figure A4.2 c) and the attenuation values bears little resemblance to the basic algorithm results (Figure A4.2 b) and Figure A4.2 d), but the tendency for same stressors to converge to similar values is apparent. The best λ decreased from 0.74 to 0.15. Analysis of λ contributions indicated that λ_X still dominated λ.

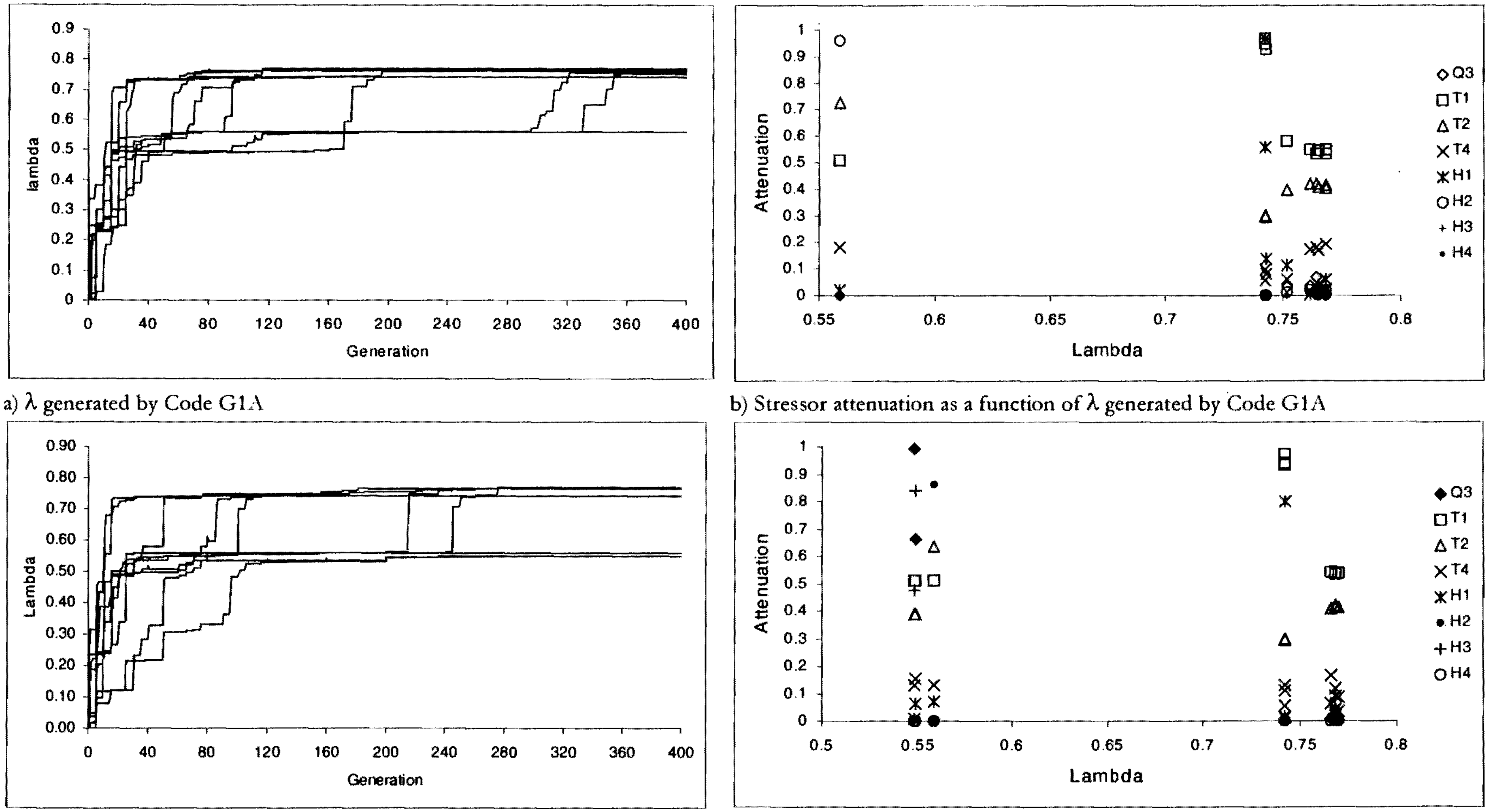
The problem might still arise that if the arithmetic average minimum λ_X is used as an aggregation measure, that some sources may have 0 acceptability while other have a high acceptability.

Addition of an overall minimum acceptability as criterion for λ (i.e. that corresponds to a conjunction of all source and stressor λ_x values) produces the results depicted in Figure A4.5. This shows that the best λ is still lower (about 0.1) and anomalous behaviour of the flow-stressor attenuation.

The apparently obvious next step, combining the average minimum λ_x aggregation without an equity constraint produced degenerate $\lambda = 0.99$ for all runs in all scenarios within no more than 80 generations. Figure A4.9 a) to g) shows the variability in the best stressor attenuation values indicating no tendency for stressor-source specific attenuation to converge (the exception being xH4, which was consistently zero).

Figure A4.6 compares the scenario where the toxic attenuation acceptability range was reduced. The attenuation values in comparison to the baseline showed the inherent danger of using average minimum aggregation. The overall λ only decreased very slightly. When using the conjunction aggregation, λ decreased to about 0.09 but when using the conjunction aggregation with equity constraints the stressor specific attenuation remained essentially the same with toxics attenuation being slightly lower. This might be an artefact of the membership function, which asymptotically approaches 0 and 1.

The impact of placing lower risk constraints on the optimal solution resulted in the data depicted in Figure 4.7. When no equity constraints were used and in the absence of conjunctive aggregation, source 1 is heavily penalised. When both types of constraints are added (Figure A4.8), λ comes down to about 0.01 with λ_x still being dominant with λ_{eq} closely following. Interestingly enough, the risk constraint (in terms of λ_R) has very little direct impact on λ.



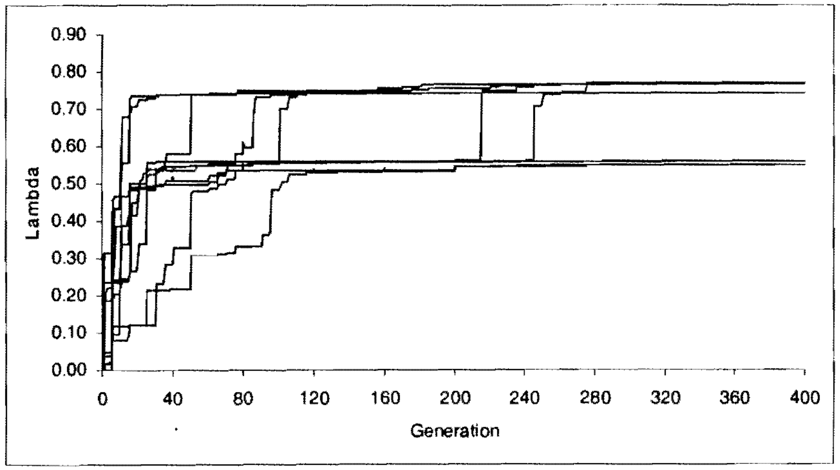
a) λ generated by Code G1A

b) Stressor attenuation as a function of λ generated by Code G1A

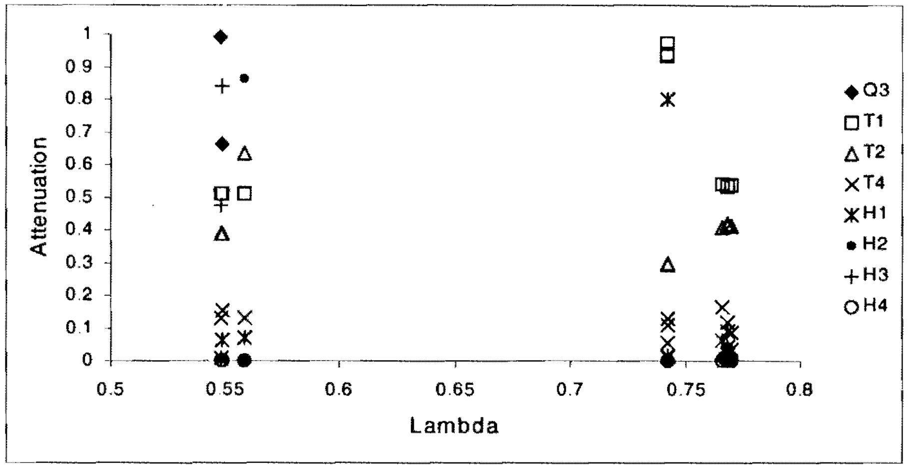
c) λ generated by code G1B

d) Stressor attenuation as a function of λ generated by Code G1B

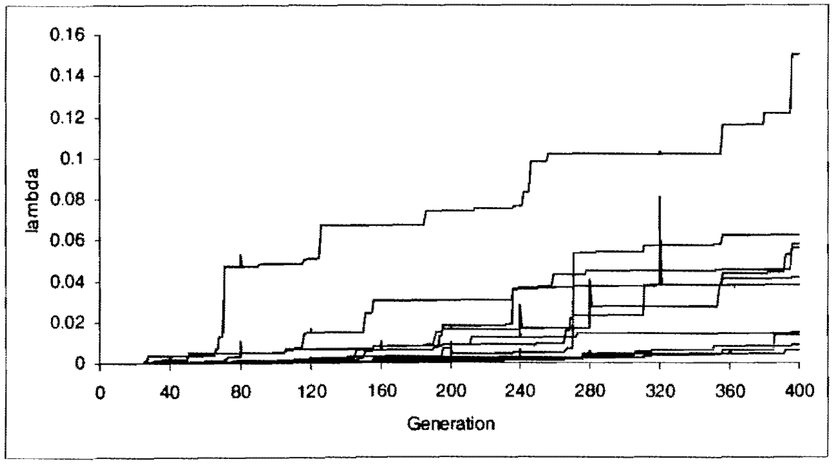
Figure A4.1 A comparison of the effect of control variable initialisation distribution on the performance of the genetic algorithm.



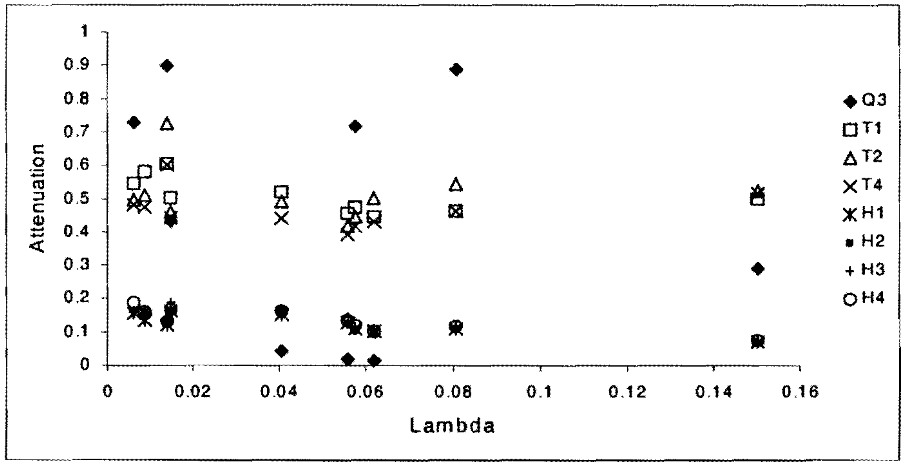
a) λ generated by Code G1B



b) Stressor attenuation as a function of λ generated by Code G1B

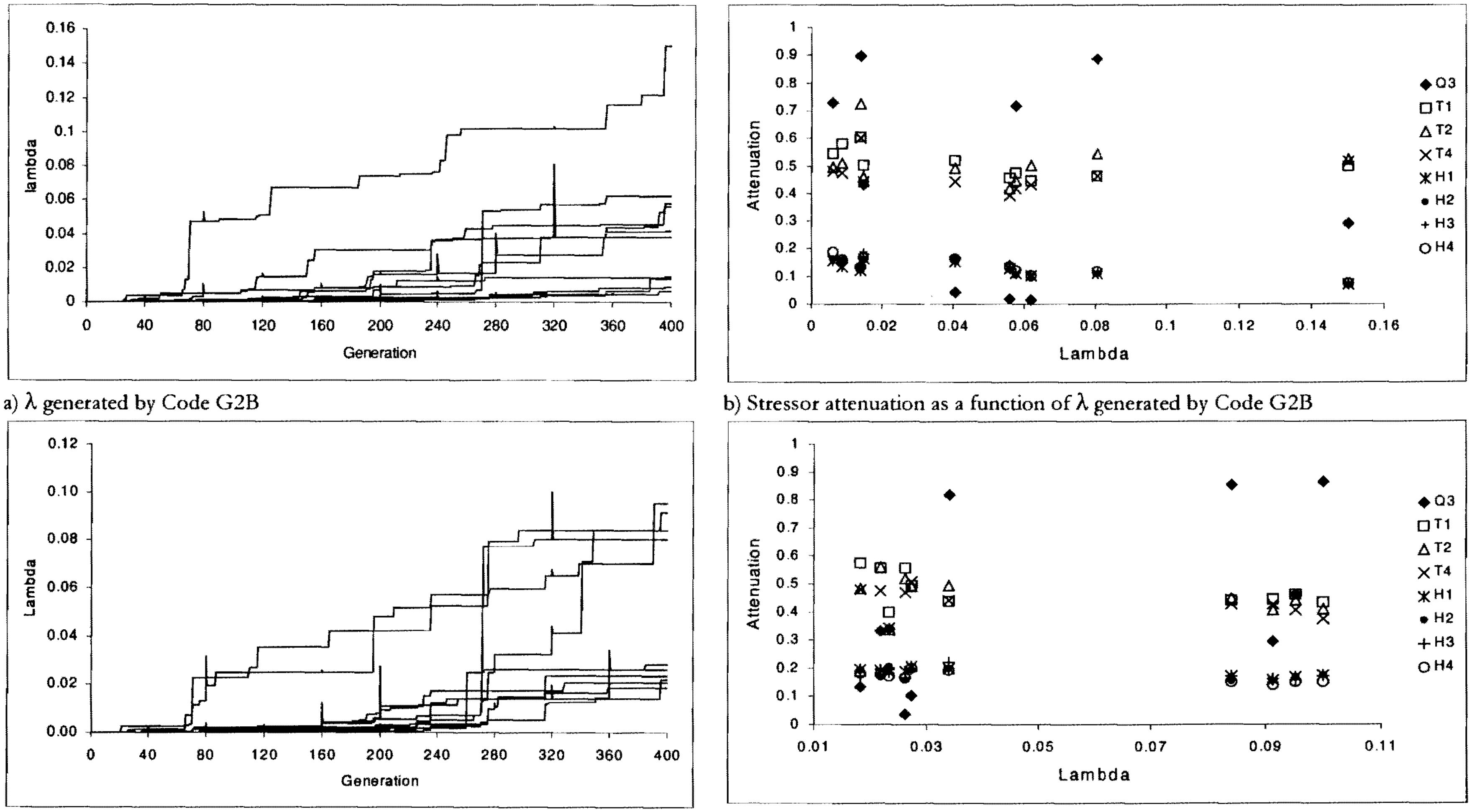


c) λ generated by code G2B



d) Stressor attenuation as a function of λ generated by Code G2B

Figure A4.2 A comparison of the effect of the addition of an equity constraint on the performance of the genetic algorithm.



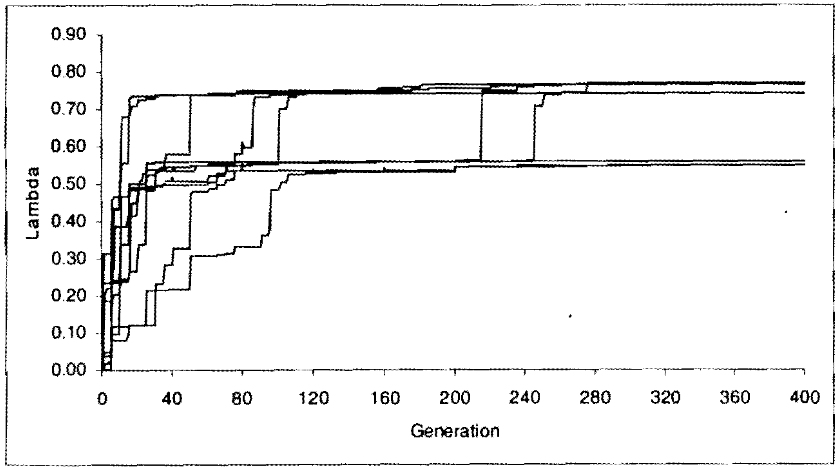
a) λ generated by Code G2B

b) Stressor attenuation as a function of λ generated by Code G2B

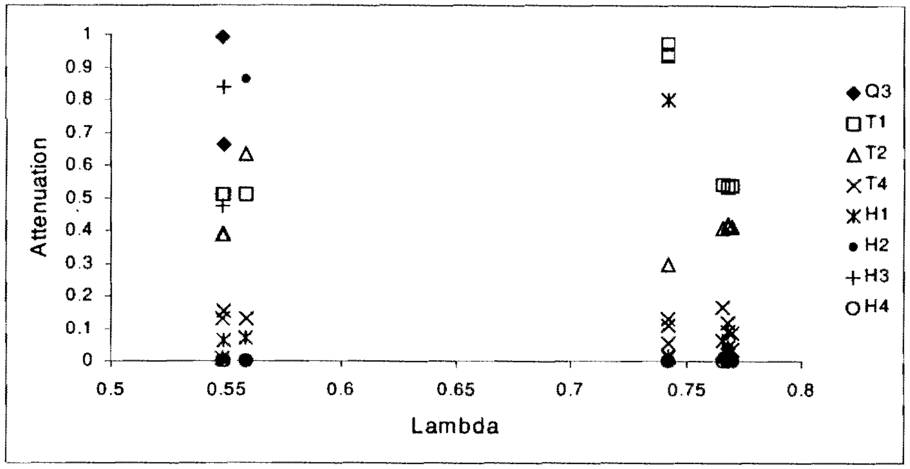
c) λ generated by code G3B

d) Stressor attenuation as a function of λ generated by Code G3B

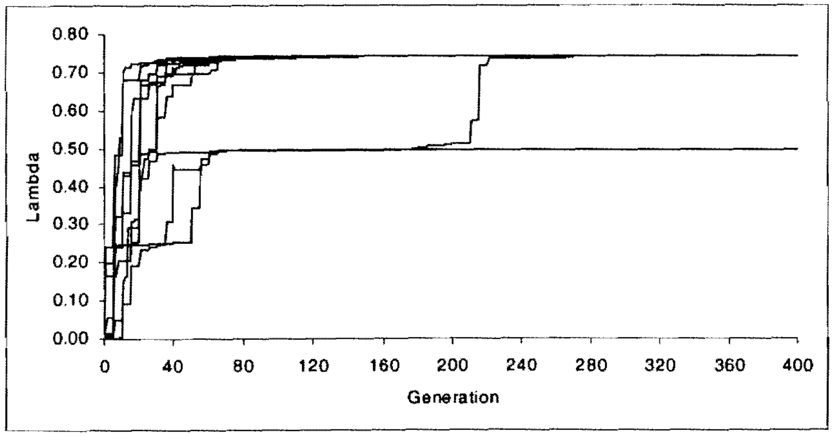
Figure A4.3 A comparison of the effect of the addition of an equity constraint and change to minimum attenuation acceptability on the performance of the genetic algorithm.



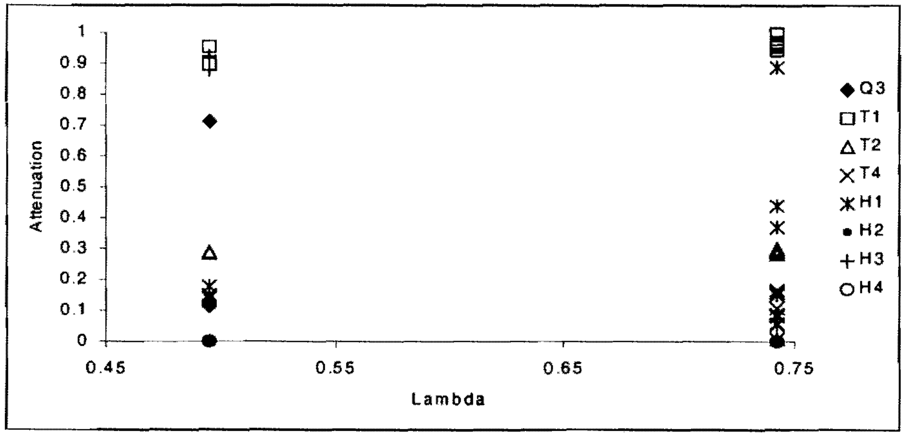
a) λ generated by Code G1B Scenario 2



b) Stressor attenuation as a function of λ generated by Code G1B Scenario 2

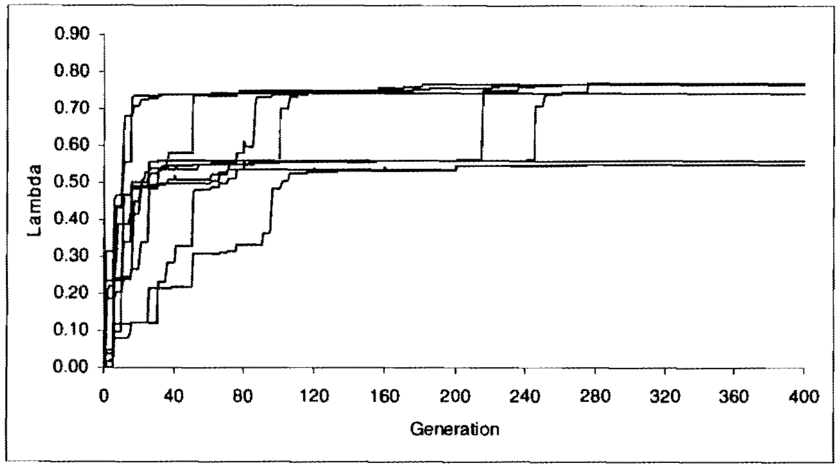


c) λ generated by code G1B Scenario 3

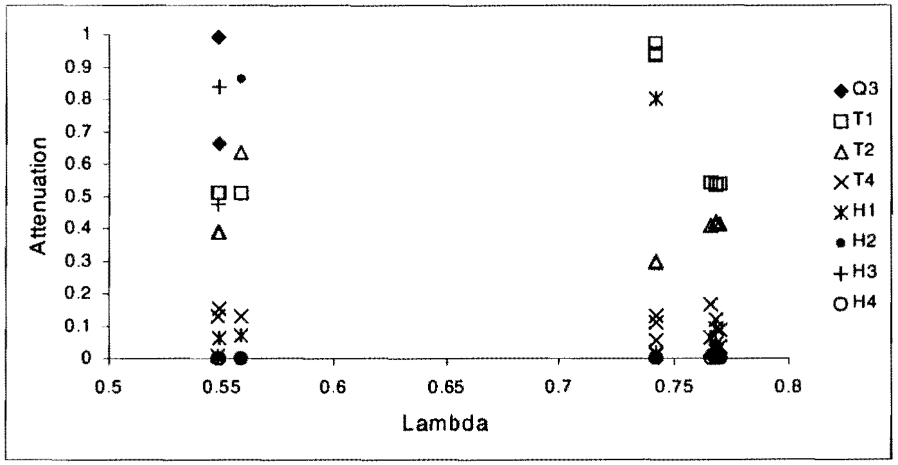


d) Stressor attenuation as a function of λ generated by Code G1B Scenario 3

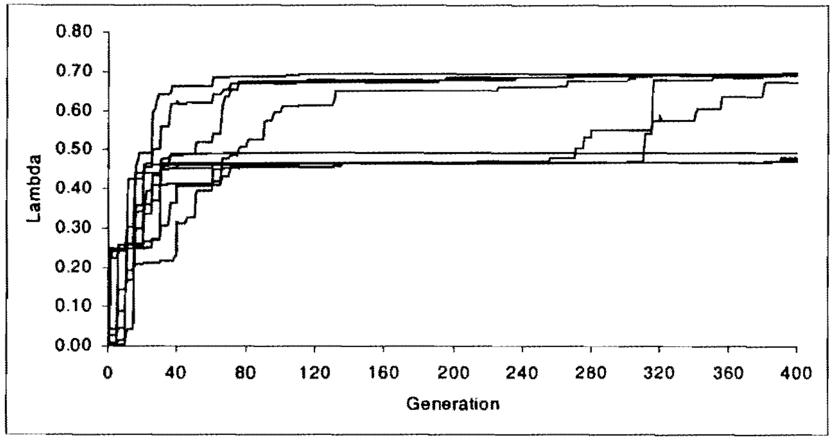
Figure A4.4 A comparison of the effect of a change of attenuation acceptability for toxics at source 1 on the performance of the genetic algorithm.



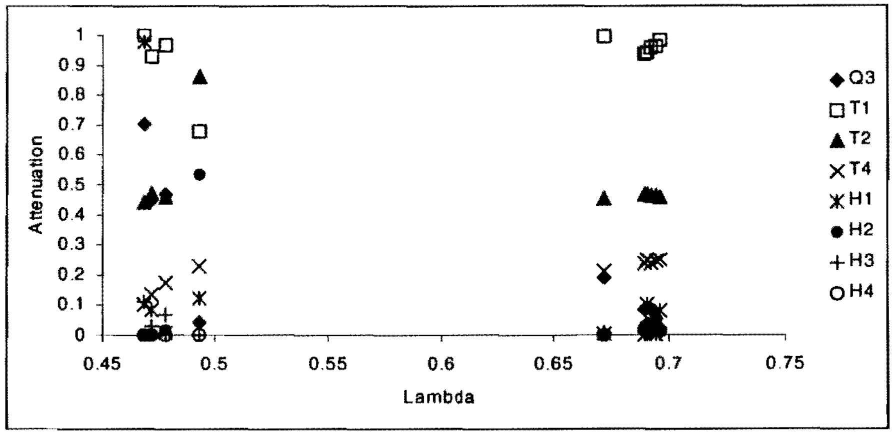
a) λ generated by Code G1B Scenario 2



b) Stressor attenuation as a function of λ generated by Code G1B Scenario 2

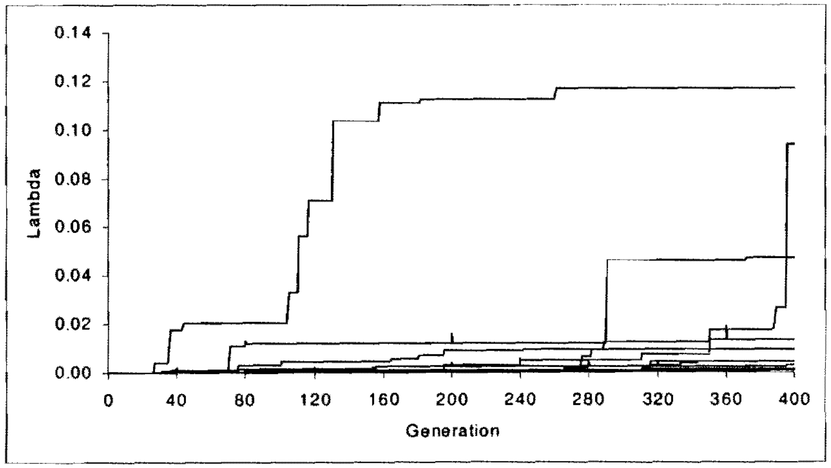


c) λ generated by code G1B Scenario4

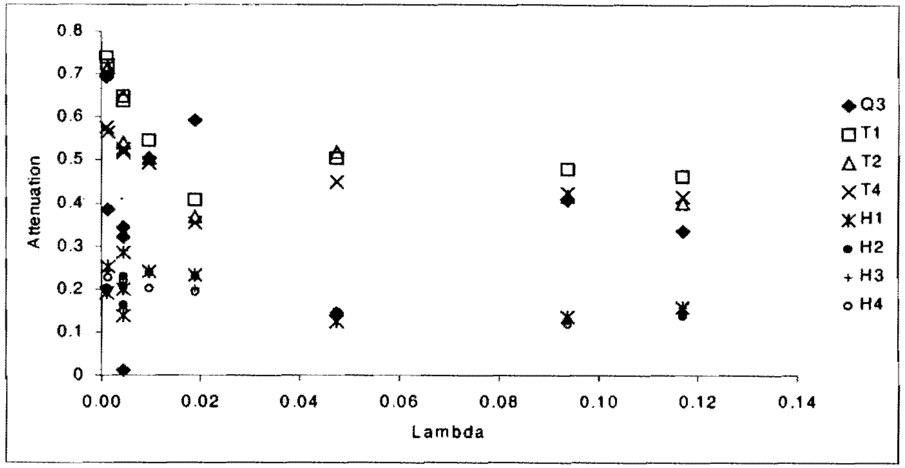


d) Stressor attenuation as a function of λ generated by Code G1B Scenario 4

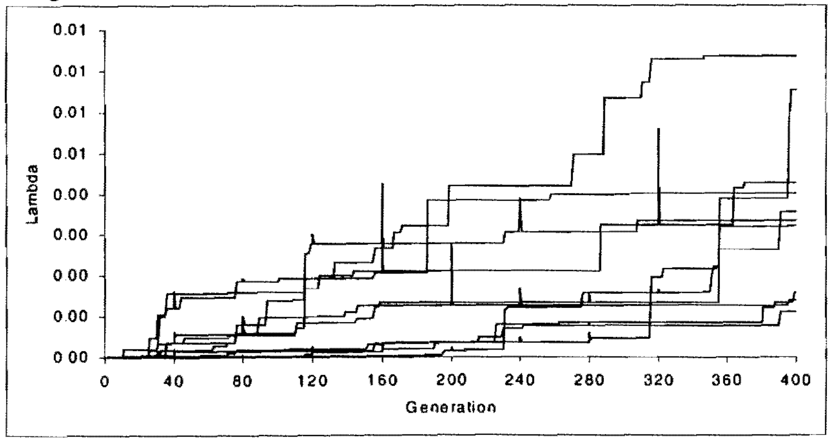
Figure A4.5 A comparison of a change in risk acceptability on the performance of the genetic algorithm.



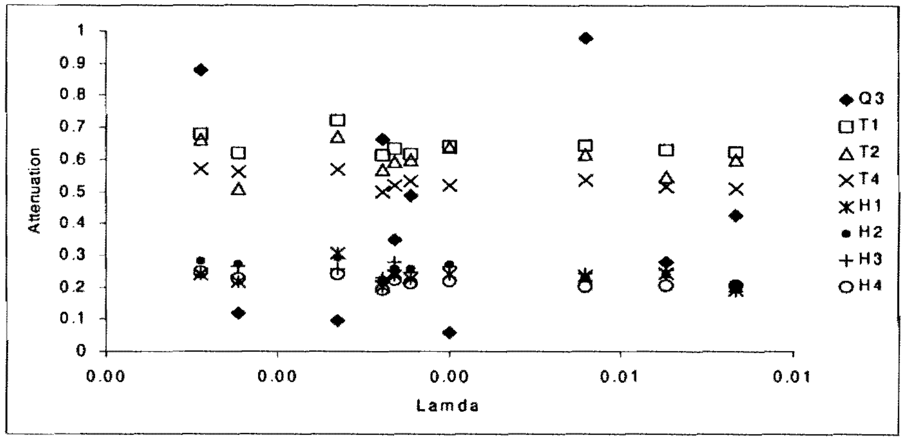
a) λ generated by Code G3B Scenario 3



b) Stressor attenuation as a function of λ generated by Code G3B Scenario 3

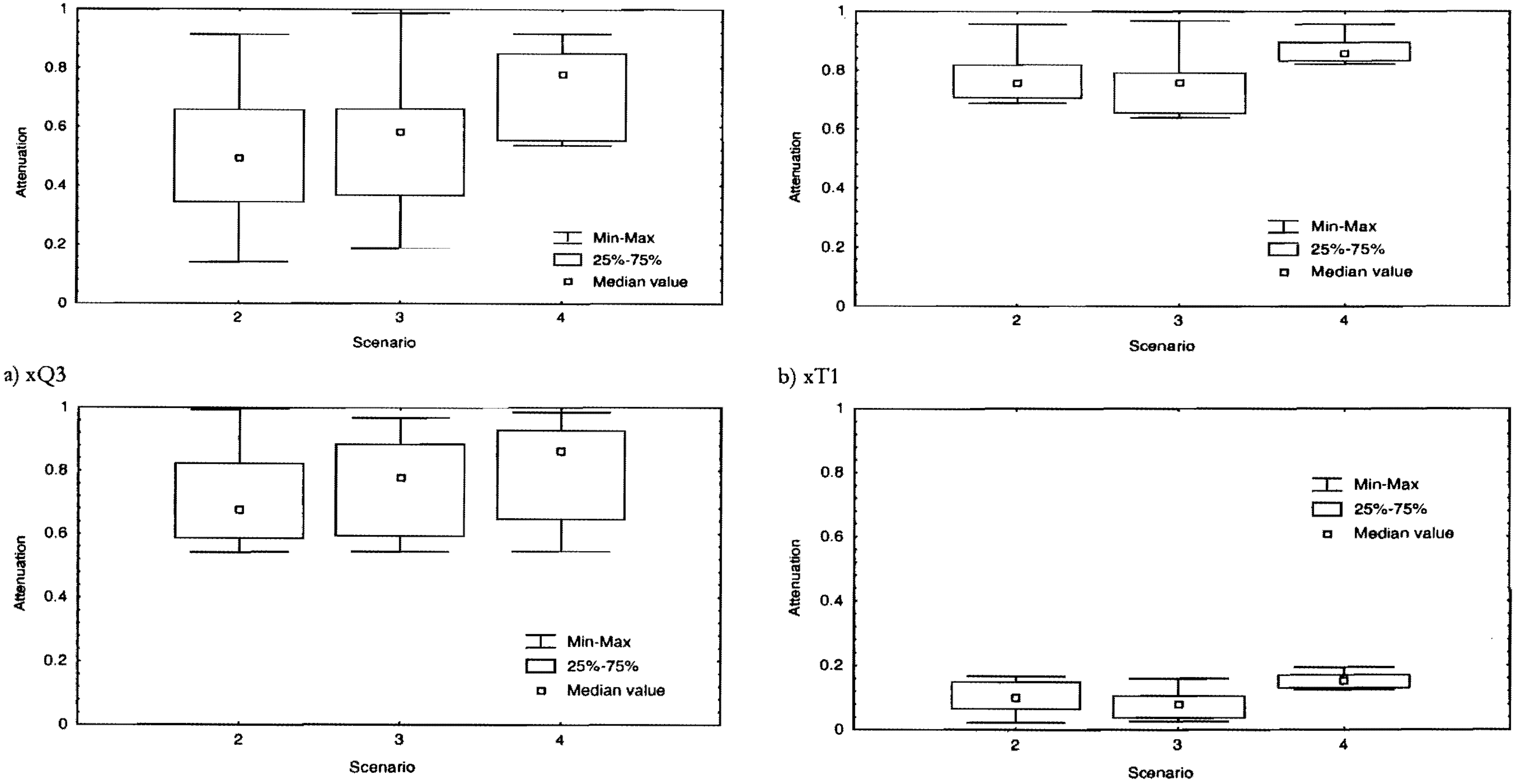


c) λ generated by code G3B Scenario 4



d) Stressor attenuation as a function of λ generated by Code G1B Scenario 4

Figure A4.6 A comparison of a change in risk acceptability on the performance of the genetic algorithm with both conjunctive aggregation and equity constraint.



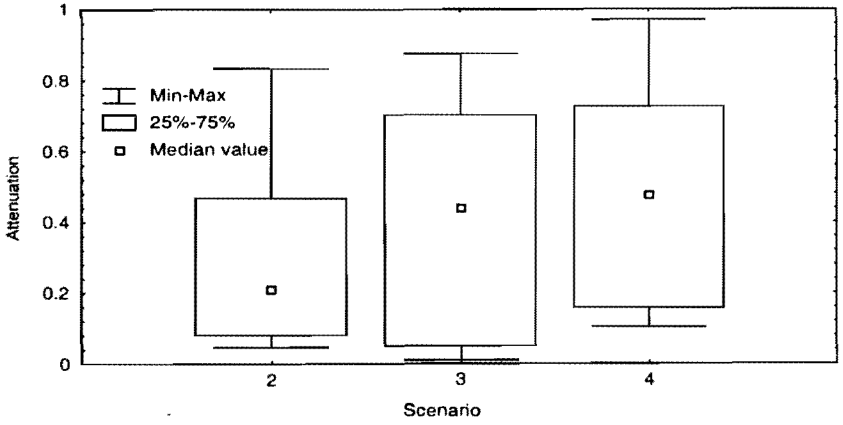
a) xQ3

b) xT1

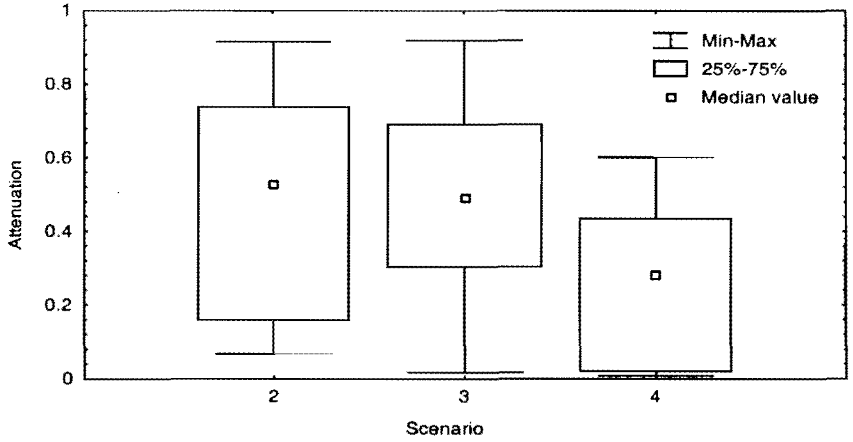
c) xT2

d) xT4

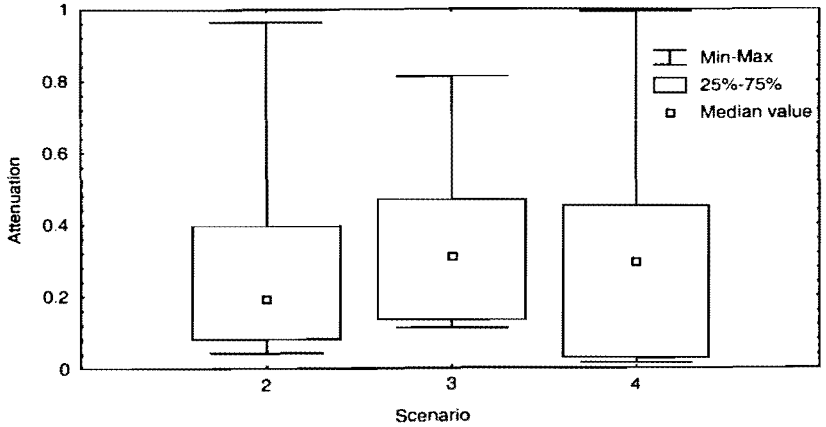
Figure A4.7 The distribution of stressor attenuation values generated by code G4B with degenerate λ = 0.99 values.



e) xH1



f) xH2



g) xH3

Figure A4.6 (continued) The distribution of stressor attenuation values generated by code G4B with degenerate $\lambda = 0.99$ values.

A4.3 THE BASIC ALGORITHM CODING G1A IN MS-QBASIC

```
DECLARE SUB climb (t%, m%, x!(), fs!(), pmin!(), pmax!())
DECLARE SUB datainput (infil$, s%, c%, s0!(), f!(), k!(), tau!(), z%(), qm!(), e!(), user!(), regl!())
DECLARE SUB ftox (s%, c%, f!(), x!())
DECLARE SUB findvalue (f!, x!(), t%)
DECLARE SUB initialize (x!(), pmin!(), pmax!(), fs!(), t%, m%)
DECLARE SUB QuickSort (ndim%, SList!(), PList!(), Left%, Right%)
DECLARE SUB encode (x!(), t%, chrom$)
DECLARE SUB offspring (m%, gen%, ch$())
DECLARE SUB decode (chrom$, t%, y!())
DECLARE SUB binadd (x$, y$, z$)
DECLARE SUB binneg (a$, c2$)
DECLARE SUB cvbin (x!, a$)
DECLARE SUB cvdec (y$, y!)
DECLARE SUB Partition (ndim%, SList!(), PList!(), Left%, Right%, part%)
DECLARE SUB calcrisk (i%, j%, muef!(), mus!(), rsk!())
DECLARE SUB value (lamda!, x!())
DECLARE SUB intadd (x!(), y!(), z!())
DECLARE SUB intdiv (x!(), y!(), z!())
DECLARE SUB intinv (x!(), z!())
DECLARE SUB intmult (x!(), y!(), z!())
DECLARE SUB linv (y!, mu!, s!, x!())
DECLARE SUB mueff (i%, j%, e!(), s!(), muef!())
DECLARE SUB mustres (i%, j%, a!, qm!(), st!(), mus!(), poss!)
DECLARE SUB ninv (y!, mu!, s!, x!())
DECLARE SUB satisfy (s%, c%, user!(), regl!(), maxr!, f!(), lamda!)
DECLARE SUB stresdist (a%, s%, c%, p!(), k!(), f!(), tau!(), z%(), s!(), a!)
DECLARE SUB tfnalfa (a!(), alfa!, a1!, a2!)
DECLARE SUB xtof (s%, c%, f!(), x!())
CONST pi = 3.1415926536#
s% = 3  'Number of stressors
c% = 4  'Number of sources
n% = 20 'Number of confidence levels
p% = 10             'epoch number
eps = .0001
CLS
RANDOMIZE TIMER
DIM s0(s%, c% + 1, 3), f(s%, c%), e(s%, 2), k(s%, c%), tau(c% + 1), qm(c%, 2), user(s%, c%, 2),
regl(2), z%(c% + 1), xf(s% * c%)
DEF fnmustepup (min, max, x)
        IF x <= min THEN
                fnmustepup = 0
        ELSEIF x >= max THEN
                fnmustepup = 1
        ELSE
                fnmustepup = (x - min) / (max - min)
        END IF
END DEF
DEF fnmustepdown (min, max, x)
        IF x <= min THEN
                fnmustepdown = 1
        ELSEIF x >= max THEN
                fnmustepdown = 0
        ELSE
                fnmustepdown = (max - x) / (max - min)
        END IF
END DEF
DEF fnsatisfy (x1, x2, x)
```

```
        yl = .99
        y2 = .01
        k = LOG(y2 * (1 - yl) / ((1 - y2) * yl)) / (xl - x2)
        ax = EXP(LOG(yl / (1 - yl)) + k * xl)
            fnsatisfy = ax * EXP(-k * x) / (1 + ax * EXP(-k * x))
END DEF
DEF fntriang (a, b, c, x)
        IF x < a OR x > c THEN
            fntriang = 0
        ELSEIF x <= b THEN
            fntriang = (x - a) / (b - a)
        ELSE
            fntriang = (c - x) / (c - b)
        END IF
END DEF

DEF fnmin (a, b)
        IF a <= b THEN fnmin = a ELSE fnmin = b
END DEF
DEF fnmax (a, b)
        IF a <= b THEN fnmax = b ELSE fnmax = a
END DEF
DEF fnnorm (x, mu, s)
        fnnorm = EXP(-(x - mu) ^ 2 / (2 * s ^ 2)) / (2 * SQR(2 * pi))
END DEF
DEF fnlognorm (x, mu, s)
        fnlognorm = EXP(-(LOG(x) - mu) ^ 2 / (2 * s * SQR(2 * pi))) / (x * s * SQR(2 * pi))
END DEF
'------------Inputs--------------------------
t% = 8: m% = 2 * t%
DIM x(m%, t%), xi(t%), y(m%, t%), yi(t%), ch$(m%), fs(m%), xb(n%, t%)
DIM oldx(2, t%), lr(m%), lx(m%)
DIM sumxb(t%), xbmax(t%), xbmin(t%), oldxbmax(t%), oldxbmin(t%)
DIM SList(m%), PList(m%, t%)
fil$ = "gla": f$ = "f.txt": x$ = "x.txt"
idir$ = "c:\data\optin": iex$ = ".dat"
odir$ = "c:\data\"
FOR filecount% = 1 TO 3
        c$ = RIGHT$(STR$(filecount% + 1), 1)
        infil$ = idir$ + c$ + iex$
        outfil1$ = odir$ + fil$ + c$ + f$
        outfil2$ = odir$ + fil$ + c$ + x$
        CALL datainput(infil$, s%, c%, s0(), f!(), k!(), tau!(), z%(), qm(), e!(), user!(), regl())
        CALL ftox(s%, c%, f(), xf())
        m% = 2 * t%
        FOR i% = 1 TO t%
            vbestx(i%) = 0
        NEXT
        REDIM x(m%, t%), xi(t%), y(m%, t%), yi(t%), ch$(m%), fs(m%), xb(n%, t%)
        REDIM oldx(2, t%), lr(m%), lx(m%)
        REDIM sumxb(t%), xbmax(t%), xbmin(t%), oldxbmax(t%), oldxbmin(t%)
        REDIM SList(m%), PList(m%, t%)
        OPEN outfil1$ FOR OUTPUT AS #5
        OPEN outfil2$ FOR APPEND AS #6
'======================OPTIMIZATION BY GENETIC LGORITHM================
'                 =============MAIN PROGRAMME=====================
        try% = 0: scount% = 0
        DO
        try% = try% + 1
        PRINT try%;
```

```
outloop% = 0
vbestf = 1000
FOR j% = 1 TO 2    '--Find 1st two suitable values as parents
        DO
                FOR i% = 1 TO t%
                        xi(i%) = RND
                        x(j%, i%) = xi(i%)
                NEXT
                CALL findvalue(f, xi(), t%)
                fs(j%) = f
        LOOP UNTIL f < 1
NEXT                    '--Arrange 1st 2 values--
IF fs(2) < fs(1) THEN
        SWAP fs(1), fs(2)
        FOR i% = 1 TO t%
                SWAP x(1, i%), x(2, i%)
        NEXT
END IF          '---------------------------
DO
        FOR i% = 1 TO t%
                pmin(i%) = 0
                pmax(i%) = 1
                shift(i%) = 0
                bestx(i%) = 0
        NEXT
        outloop% = outloop% + 1          '--Prepare for epoch---
        count% = 0: gen% = 0: bestf = 100000
        sgen% = 0
        FOR i% = 1 TO t%
                sumxb(i%) = 0
                xbmax(i%) = 0
                xbmin(i%) = 9999
        NEXT
        CALL initialize(x(), pmin(), pmax(), fs(), t%, m%)'----------------------
        CALL QuickSort(t%, fs(), x(), 1, m%)
        FOR i% = 1 TO t%
                x0(i%) = x(1, i%)
        NEXT
        CALL findvalue(f, x0(), t%)
        PRINT #5, scount%;
        FOR i% = 1 TO t%
                PRINT #5, x(1, i%);
        NEXT
        PRINT #5, fs(1); lamdar; lamdax
        DO                      '--Start epoch----------------
                sgen% = sgen% + 1
                count% = count% + 1
                scount% = scount% + 1
                bestf1 = fnmin(bestf, fs(1))
                IF bestf1 < bestf THEN
                        FOR i% = 1 TO t%
                                bestx(i%) = x(1, i%)
                        NEXT
                        bestf = bestf1
                END IF
                FOR i% = 1 TO m%          '---produce chromosomes---
                        FOR j% = 1 TO t%
                                xi(j%) = x(i%, j%)
                        NEXT
                        c$ = ""
```

```
        CALL encode(xi(), t%, c$)
        ch$(i%) = c$
NEXT
CALL offspring(m%, gen%, ch$()) '---do genetic manipulations
FOR i% = 1 TO m%
        c$ = ch$(i%)
        CALL decode(c$, t%, xi())
        FOR j% = 1 TO t%
                x(i%, j%) = xi(j%)
        NEXT
        CALL findvalue(f, xi(), t%)
        fs(i%) = f
NEXT
CALL QuickSort(t%, fs(), x(), 1, m%)
FOR i% = 1 TO t%
        x0(i%) = x(1, i%)
NEXT
CALL findvalue(f, x0(), t%)
PRINT #5, scount%;
FOR i% = 1 TO t%
        PRINT #5, x(1, i%);
NEXT
PRINT #5, fs(1); lamdar; lamdax
IF count% = 1 THEN              '---prepare for next epoch
        FOR i% = 1 TO t%         '--initialise max-min calc params
                oldxbmax(i%) = x(1, i%)
                oldxbmin(i%) = x(1, i%)
        NEXT
ELSE
        FOR i% = 1 TO t%
                oldxbmax(i%) = xbmax(i%)
                oldxbmin(i%) = xbmin(i%)
        NEXT
END IF
IF sgen% < 5 THEN
        FOR i% = 1 TO t%
                sumxb(i%) = sumxb(i%) + x(1, i%)
                newxb = x(1, i%)
                oldxbmax = oldxbmax(i%): oldxbmin = oldxbmin(i%)
                xbmax(i%) = fnmax(oldxbmax, newxb)
                xbmin(i%) = fnmin(oldxbmin, newxb)
        NEXT
ELSE
        FOR i% = 1 TO t%
                sumxb(i%) = sumxb(i%) + x(1, i%)
                rl = 2 * (xbmax(i%) - xbmin(i%))
                IF rl < .4 THEN
                        rl = .4
                ELSEIF rl > .5 THEN
                        rl = .5
                END IF
                pmax(i%) = x(1, i%) + rl * (pmax(i%) - pmin(i%))
                pmin(i%) = x(1, i%) - rl * (pmax(i%) - pmin(i%))
                shift(i%) = ((sumxb(i%) / sgen%) - .5 * (pmax(i%) + pmin(i%))) /
(pmax(i%) - pmin(i%))
                pmax(i%) = pmax(i%) + shift(i%) * (pmax(i%) - pmin(i%))
                IF pmax(i%) >= 1 THEN pmax(i%) = .99999
                pmin(i%) = pmin(i%) + shift(i%) * (pmax(i%) - pmin(i%))
                IF pmin(i%) <= 0 THEN pmin(i%) = .00001
        NEXT
```

```
                    CALL climb(t%, m%, x(), fs(), pmin(), pmax())
                    sgen% = 0
                    FOR i% = 1 TO t%
                        sumxb(i%) = 0
                    NEXT
                END IF
                CALL QuickSort(t%, fs(), x(), 1, m%)
            LOOP UNTIL count% = 40
            IF bestf < vbestf THEN
                FOR i% = 1 TO t%
                    vbestx(i%) = bestx(i%)
                NEXT
                vbestf = bestf
                vbestlr = bestlr: vbestlx = bestlx
            END IF
            FOR i% = 1 TO t%
                x(1, i%) = bestx(i%)
            NEXT
        LOOP UNTIL outloop% = p%
        PRINT
        FOR i% = 1 TO t%
            PRINT vbestx(i%);
            PRINT #6, vbestx(i%);
        NEXT
        PRINT #6, vbestf
        PRINT vbestf
        LOOP UNTIL try% = 10
        CLOSE #5: CLOSE #6
NEXT 'filecount%
'=======================END OF MAIN PROGRAMME=======================

SUB binadd (x$, y$, z$)
z$ = "": co = 0
FOR i% = 16 TO 1 STEP -1
    a = VAL(MID$(x$, i%, 1)): b = VAL(MID$(y$, i%, 1))
    c = a + b + co
    IF c >= 2 THEN
        d = 2 - c
        co = 1
    ELSE
        d = c
        co = 0
    END IF
    z$ = RIGHT$(STR$(d), 1) + z$
NEXT
END SUB

SUB binneg (a$, c2$)
CALL cvbin(1, one$)
FOR i% = 1 TO 15
    one$ = "0" + one$
NEXT
c$ = ""
FOR i% = 1 TO 16
    IF MID$(a$, i%, 1) = "1" THEN
        c$ = c$ + "0"
    ELSEIF MID$(a$, i%, 1) = "0" THEN
        c$ = c$ + "1"
    END IF
NEXT
```

```
CALL binadd(c$, one$, c2$)
END SUB

SUB calcrisk (i%, j%, muef(), mus(), rsk())
        maxr = 0: mx = 0
        muef1 = muef(i%, j%, 1): muef2 = muef(i%, j%, 2)
        mus1 = mus(i%, j%, 1): mus2 = mus(i%, j%, 2)
        rsk(i%, j%, 1) = fnmin(muef1, mus1)
        rsk(i%, j%, 2) = fnmin(muef2, mus2)
END SUB

SUB climb (t%, m%, x(), fs(), pmin(), pmax())
DIM range(t%), xi(t%)
        FOR i% = 1 TO t%
                range(i%) = pmax(i%) - pmin(i%)
        NEXT
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        x(i%, j%) = RND * range(j%) + pmin(j%)
                        xi(j%) = x(i%, j%)
                NEXT
                CALL findvalue(f, xi(), t%)
                fs(i%) = f
        NEXT
END SUB

SUB cvbin (x, a$)
a$ = ""
IF x >= 0 THEN
        xa = x
        FOR i% = 16 TO 1 STEP -1
                a = 2 ^ (i% - 1)
                IF a > xa THEN
                        p$ = "0"
                ELSE
                        p$ = "1"
                        xa = xa - a
                END IF
                a$ = a$ + p$
        NEXT
ELSE
        y = -x
        ya = y
        FOR i% = 16 TO 1 STEP -1
                a = 2 ^ (i% - 1)
                IF a > ya THEN
                        p$ = "0"
                ELSE
                        p$ = "1"
                        ya = ya - a
                END IF
                a$ = a$ + p$
        NEXT
        CALL binneg(a$, c$)
        a$ = c$
END IF
END SUB

SUB cvdec (y$, y)
        y = 0
```

```
        FOR i% = 1 TO 16
                y = y + VAL(MID$(y$, i%, 1)) * 2 ^ (16 - i%)
        NEXT
END SUB

SUB datainput (infil$, s%, c%, s0!(), f!(), k!(), tau!(), z%(), qm(), e!(), user!(), regl())
OPEN infil$ FOR INPUT AS #1
ct% = 0
FOR i% = 1 TO s%
        FOR k% = 1 TO 3
                FOR j% = 0 TO c%
                        INPUT #1, s0(i%, j%, k%): ct% = ct% + 1
                NEXT
        NEXT
NEXT
FOR j% = 1 TO 2
        FOR i% = 1 TO c%
                INPUT #1, qm(i%, j%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO s%
        FOR k% = 1 TO 2
                FOR j% = 1 TO c%
                        INPUT #1, e(i%, k%): ct% = ct% + 1
                NEXT
        NEXT
NEXT
FOR j% = 1 TO s%
        FOR i% = 1 TO c%
                INPUT #1, f(j%, i%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO c%
        INPUT #1, z%(i%): ct% = ct% + 1
NEXT
FOR j% = 1 TO s%
        FOR i% = 1 TO c%
                INPUT #1, k(j%, i%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO c%
        INPUT #1, tau(i%): ct% = ct% + 1
NEXT
FOR k% = 1 TO s%
FOR j% = 1 TO 2
        FOR i% = 1 TO c%
                INPUT #1, user(k%, i%, j%): ct% = ct% + 1
        NEXT
NEXT
NEXT
INPUT #1, regl(1)
INPUT #1, regl(2)
CLOSE #1
END SUB

SUB decode (chrom$, t%, y())
'-----------------------------Decode chromosome-----------------
DIM y$(t%)
FOR i% = 1 TO t%
```

```
        p% = 1 + 16 * (i% - 1)
        y$(i%) = MID$(chrom$, p%, 16)
        y$ = y$(i%)
        IF VAL(LEFT$(y$, 1)) = 1 THEN
                CALL binneg(y$, y1$)
                CALL cvdec(y1$, y)
                y = -y
        ELSE
                CALL cvdec(y$, y)
        END IF
        y(i%) = y / 1000
NEXT
END SUB

SUB discrete (snum%, num%, xlow, xup, x())
FOR i% = 1 TO num%
        x(snum%, i%) = xlow + (i% - 1) * (xup - xlow) / (num% - 1)
NEXT
END SUB

SUB encode (x(), t%, chrom$)
'----------------------------Encode chromosome; 3 decimal accuracy---
chrom$ = ""
FOR i% = 1 TO t%
        x = x(i%) * 1000
        CALL cvbin(x, a$)
        chrom$ = chrom$ + a$
NEXT
END SUB

SUB findvalue (f, x(), t%)
        SHARED s%, c%
        er% = 0
        FOR i% = 1 TO t%
                IF x(i%) < 0 OR x(i%) > 1 THEN er% = 1
        NEXT
        IF er% = 0 THEN
                CALL value(lamda, x())
                f = 1 - lamda
        ELSE
                f = 101010
        END IF
END SUB

SUB ftox (s%, c%, f(), x())
SHARED z%(), t%
k% = 0
FOR i% = 1 TO s%
        FOR j% = 1 TO c%
                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) = 0) OR i% = s%
THEN
                k% = k% + 1
                x(k%) = f(i%, j%)
                END IF
        NEXT
NEXT
IF t% <> k% THEN t% = k%
END SUB

SUB initialize (x(), pmin(), pmax(), fs(), t%, m%)
```

```
'------------------------------------Initialize variables-----
        DIM xi(t%)
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        pwr% = INT(RND * 2) + 1
                        x(i%, j%) = x(1, j%) + (-1) ^ pwr% * RND * .5 * (pmax(j%) - pmin(j%))
                        xi(j%) = x(i%, j%)
                NEXT
                        CALL findvalue(f, xi(), t%)
                        fs(i%) = f
        NEXT
END SUB

SUB intadd (x(), y(), z())
z(1) = x(1) + y(1)
z(2) = x(2) + y(2)
END SUB

SUB intdiv (x(), y(), z())
DIM a(2)
CALL intinv(y(), a())
temp = a(1)
a(1) = a(2)
a(2) = temp
CALL intmult(x(), a(), z())
END SUB

SUB intinv (x(), z())
z(1) = 1 / x(2): z(2) = 1 / x(1)
END SUB

SUB intmult (x(), y(), z())
'a = x(1) * y(1): b = x(1) * y(2): c = x(2) * y(1): d = x(2) * y(2)
z(1) = x(1) * y(1)'fnmin(d, fnmin(c, fnmin(a, b)))
z(2) = x(2) * y(2)'fnmax(d, fnmax(c, fnmax(a, b)))
END SUB

SUB intsub (x(), y(), z())
z(1) = x(1) - y(2)
z(2) = x(2) - y(1)
END SUB

SUB linv (y, mu, s, x())
        y1 = y * .99999 / SQR(2 * pi)
        zpos = SQR(-2 * LOG(ABS(SQR(2 * pi) * y1)))
        zneg = -SQR(-2 * LOG(ABS(SQR(2 * pi) * y1)))
        x(1) = EXP(mu1 + s * zneg): x(2) = EXP(mu1 + s * zpos)
END SUB

SUB mueff (i%, j%, e(), s(), muef())
        IF i% = 1 THEN
                a = .2: b = .8
                q01 = s(1, 0, 1): IF q01 = 0 THEN q01 = .0001
                q02 = s(1, 0, 2): IF q02 = 0 THEN q02 = .0001
                x1 = (q01 - s(1, j%, 1)) / q01: x2 = (q02 - s(1, j%, 2)) / q02
        ELSE
                a = e(i%, 1): b = e(i%, 2)
                x1 = s(i%, j%, 1): x2 = s(i%, j%, 2)
        END IF
        e1 = fnmustepup(a, b, x1)
```

```
        e2 = fnmustepup(a, b, x2)
        muef(i%, j%, 1) = e1
        muef(i%, j%, 2) = e2
END SUB

SUB mustres (i%, j%, a, qm(), st(), mus(), poss)
                mus(i%, j%, 1) = a: mus(i%, j%, 2) = a: poss = a
END SUB

SUB ninv (y, mu, s, x())
        y1 = y / SQR(2 * pi)
        zpos = SQR(-2 * s ^ 2 * LOG(y))
        zneg = -SQR(-2 * s ^ 2 * LOG(y))
        x(1) = mu + s * zneg: x(2) = mu + s * zpos
END SUB

SUB offspring (m%, gen%, ch$())
'----------------Produce offspring-----
SHARED t%
DIM f2$(m%)
lamda = 1
'---select parents
FOR i% = 5 TO m%
DO
        pno1% = INT(-LOG(1 - RND) / lamda + 1)
        pno2% = INT(-LOG(1 - RND) / lamda + 1)
LOOP UNTIL pno1% <> pno2% AND pno1% < m% AND pno2% < m%
        f2$(i%) = ""
        dch$ = ""
        FOR j% = 1 TO t%
                gen% = gen% + 1
                byte% = 16 * (j% - 1) + 1
                slct% = INT(RND * 2)      'randomly select parent 1 or 2
                                IF slct% = 1 THEN
                        a$ = MID$(ch$(pno1%), byte%, 16)
                ELSE
                        a$ = MID$(ch$(pno2%), byte%, 16)
                END IF
                IF gen% = 10 THEN
                        mubit1% = INT(RND * 16) + 1: mubit2% = INT(RND * 16) + 1
                        dummy$ = ""
                        FOR k% = 1 TO 16
                                IF k% <> mubit1% OR k% <> mubit2% THEN
                                        dummy$ = dummy$ + MID$(a$, k%, 1)
                                ELSE
                                IF MID$(a$, k%, 1) = "1" THEN
                                        dummy$ = dummy$ + "0"
                                ELSE
                                        dummy$ = dummy$ + "1"
                                END IF
                                END IF
                        NEXT
                        gen% = 0
                        a$ = dummy$
                END IF
                dch$ = dch$ + a$
        NEXT j%
        f2$(i%) = dch$
NEXT i%
FOR i% = 5 TO m%
```

```
        ch$(i%) = f2$(i%)
NEXT
END SUB

SUB Partition (ndim%, SList(), PList(), Left%, Right%, part%)
        DIM temp(ndim%)
        v = SList(Right%)
        indx% = Left% - 1
        Jndx% = Right%
          DO
                DO
                  indx% = indx% + 1
                LOOP UNTIL SList(indx%) >= v
                DO
                  Jndx% = Jndx% - 1
                LOOP UNTIL SList(Jndx%) <= v
                temp = SList(indx%)
                SList(indx%) = SList(Jndx%)
                SList(Jndx%) = temp
                FOR i% = 1 TO ndim%
                temp(i%) = PList(indx%, i%)
                PList(indx%, i%) = PList(Jndx%, i%)
                PList(Jndx%, i%) = temp(i%)
                NEXT
          LOOP UNTIL Jndx% <= indx%
        SList(Jndx%) = SList(indx%)
        SList(indx%) = SList(Right%)
        SList(Right%) = temp
          FOR i% = 1 TO ndim%
                PList(Jndx%, i%) = PList(indx%, i%)
                PList(indx%, i%) = PList(Right%, i%)
                PList(Right%, i%) = temp(i%)
          NEXT
          part% = indx%
END SUB

SUB QuickSort (ndim%, SList(), PList(), Left%, Right%)
        IF Left% <= Right% THEN
                CALL Partition(ndim%, SList(), PList(), Left%, Right%, indx%)
                CALL QuickSort(ndim%, SList(), PList(), Left%, indx% - 1)
                CALL QuickSort(ndim%, SList(), PList(), indx% + 1, Right%)
        END IF
END SUB

SUB satisfy (s%, c%, user(), regl(), maxr, f(), lamda)
SHARED t%, z%(), lamdar, lamdax
min = regl(1): max = regl(2)
lamdar = fnsatisfy(min, max, maxr)
'---calculate user satisfaction---
lmdx = 0
FOR i% = 1 TO c%
        lamdai = 1
        FOR j% = 1 TO s%
                IF (j% = 1 AND z%(i%) = 1) OR (j% > 1 AND j% < s% AND z%(i%) = 0) OR j% = s%
                THEN
                        min = user(j%, i%, 1): max = user(j%, i%, 2)
                        v = f(j%, i%)
                        lx = fnsatisfy(min, max, v)
                        lamdai = fnmin(lx, lamdai)
                END IF
```

```
        NEXT
        lmdx = lmdx + lamdai
NEXT
lamdax = lmdx / c%
lamda = fnmin(lamdar, lamdax)
PRINT lamdar; lamdax,
END SUB

SUB stresdist (a%, s%, c%, p(), k(), f(), tau(), z%(), s(), a)
        SHARED n%
        DIM s0(s%, c%, 2), s1(s%, c%, 2), sv(2), qu(2), qi(2), z(2)
        DIM su(2), si(2), lu(2), li(2), qt(2), lt(2), tri(3)
        tau(0) = 0
        tau = 0
        IF a = 0 THEN a = .01
        FOR j% = 1 TO s% - 1
                mu = p(j%, 0, 1): s = p(j%, 0, 2)
                IF j% = 1 THEN
                        mu = LOG(mu)
                        CALL linv(a, mu, s, sv())
                ELSE
                        CALL ninv(a, mu, s, sv())
                END IF
                s(j%, 0, 1) = sv(1): s(j%, 0, 2) = sv(2)
                s0(j%, 0, 1) = sv(1): s0(j%, 0, 2) = sv(2)
                s1(j%, 0, 1) = sv(1): s1(j%, 0, 2) = sv(2)
        NEXT
        FOR src% = 1 TO c%
                mu = p(1, src%, 1): s = p(1, src%, 2)
                CALL linv(a, mu, s, sv())
                s0(1, src%, 1) = sv(1)
                s0(1, src%, 2) = sv(2)
                tau = tau(src%) + tau
                f = (1 - f(1, src%))
                s1(1, src%, 1) = s0(1, src%, 1) * (-f) ^ z%(src%)
                s1(1, src%, 2) = s0(1, src%, 2) * (-f) ^ z%(src%)
                qu(1) = s(1, src% - 1, 1)
                qu(2) = s(1, src% - 1, 2)
                qi(1) = s1(1, src%, 1)
                qi(2) = s1(1, src%, 2)
                FOR stres% = 2 TO s% - 1
                        degfactor = EXP(-k(stres%, src%) * tau)
                        mu = p(stres%, src%, 1): s = p(stres%, src%, 2)
                        CALL ninv(a, mu, s, sv())
                        s0(stres%, src%, 1) = sv(1)
                        s0(stres%, src%, 2) = sv(2)
                        f = (1 - f(stres%, src%))
                        s1(stres%, src%, 1) = f * (1 - z%(src%)) * s0(stres%, src%, 1) + z%(src%) *
        degfactor * s(stres%, src% - 1, 1)
                        s1(stres%, src%, 2) = f * (1 - z%(src%)) * s0(stres%, src%, 2) + z%(src%) *
        degfactor * s(stres%, src% - 1, 2)
                        su(1) = s(stres%, src% - 1, 1) * degfactor
                        su(2) = s(stres%, src% - 1, 2) * degfactor
                        si(1) = s1(stres%, src%, 1)
                        si(2) = s1(stres%; src%, 2)
                        CALL intmult(su(), qu(), lu())
                        CALL intmult(si(), qi(), li())
                        CALL intadd(lu(), li(), lt())
                        CALL intadd(qu(), qi(), qt())
                        CALL intdiv(lt(), qt(), z())
```

```
                IF z%(src%) = 0 THEN
                        s(stres%, src%, 1) = z(1): s(stres%, src%, 2) = z(2)
                ELSE
                        s(stres%, src%, 1) = su(1): s(stres%, src%, 2) = su(2)
                END IF
                s(1, src%, 1) = qt(1): s(1, src%, 2) = qt(2)
        NEXT 'stressor
    NEXT  'source
    FOR src% = 1 TO c%
        f = (1 - f(3, src%))
        tri(1) = p(3, src%, 1) * f
        tri(2) = p(3, src%, 2) * f
        tri(3) = p(3, src%, 3) * f
        CALL tfnalfa(tri(), a, a1, a2)
        s(s%, src%, 1) = a1
        s(s%, src%, 2) = a2
    NEXT
END SUB

SUB tfnalfa (a(), alfa, a1, a2)
    a1 = a(1) + alfa * (a(2) - a(1))
    a2 = a(3) - alfa * (a(3) - a(2))
END SUB

SUB trinv (alpha, a, b, c, x())
    x(1) = alpha * (b - a) - a
    x(2) = c - alpha * (c - b)
END SUB

SUB value (lamda, x())
    SHARED s%, c%, n%, a%, s0!(), f(), k!(), tau!(), z%(), qm(), e!(), user!(), regl()
    DIM min(s%), max(s%), st(s%, c% + 1, 3), mus(s%, c%, 2), muef(s%, c%, 2)
    DIM r(s%, c%, 2)
    CALL xtof(s%, c%, f(), x())
    FOR a% = 0 TO n%
        a = a% / n%
        'PRINT #2, a; : PRINT #3, a; : PRINT #4, a;
        CALL stresdist(a%, s%, c%, s0(), k(), f(), tau(), z%(), st(), a)
        maxr = 0: minr = 0
        FOR j% = 1 TO c%
            mxr = 0: mnr = 0
            FOR i% = 1 TO s%
                CALL mustres(i%, j%, a, qm(), st(), mus(), poss)
                CALL mueff(i%, j%, e(), st(), muef())
                CALL calcrisk(i%, j%, muef(), mus(), r())
                hrsk = r(i%, j%, 2)
                lrsk = r(i%, j%, 1)
                mxr = fnmax(mxr, hrsk): mnr = fnmax(mnr, lrsk)
            NEXT
            maxr = fnmax(maxr, fnmax(mxr, mnr))
        NEXT
        'PRINT #2, " ": PRINT #3, " ": PRINT #4, " "
    NEXT
    CALL satisfy(s%, c%, user(), regl(), maxr, f(), lamda)
    CALL ftox(s%, c%, f(), x())
END SUB

SUB xtof (s%, c%, f(), x())
SHARED z%()
k% = 0
```

```
FOR i% = 1 TO s%
        FOR j% = 1 TO c%
                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) = 0) OR i% = s%
THEN
                        k% = k% + 1
                        f(i%, j%) = x(k%)
                ELSE
                        f(i%, j%) = 0
                END IF
        NEXT
NEXT
END SUB
```

A4.3.1 INITIALISATION FROM AN EXPONENTIAL DISTRIBUTION: REPLACEMENT FOR SUB INITIALISE

```
SUB initialize (x(), pmin(), pmax(), fs(), t%, m%)
'-------------Initialize variables (EXP distr)-----
        DIM xi(t%)
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        a = pmin(j%): b = pmax(j%)
                        mu = .5 * (b - a)
                        l = .69314718# / mu
                        x(i%, j%) = -LOG(1 - RND * (b - a)) / l
                        xi(j%) = x(i%, j%)
                NEXT
                        CALL findvalue(f, xi(), t%)
                        fs(i%) = f
        NEXT
END SUB
```

A4.3.2 ADDING AN EQUITY CONSTRAINT: REPLACEMENT FOR SUB FINDVALUE

```
SUB findvalue (f, x(), t%)
        SHARED s%, c%, z%(), leqmin
        er% = 0
        FOR i% = 1 TO t%
                IF x(i%) < 0 OR x(i%) > 1 THEN er% = 1
        NEXT
        IF er% = 0 THEN
                CALL value(lamda, x())
                k% = 0
                leqmin = 10
                FOR i% = 1 TO s%
                        min = 10: max = 0
                        FOR j% = 1 TO c%
                                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) =
0) OR i% = s% THEN
                                        k% = k% + 1
                                        x1 = x(k%)
                                        min = fnmin(x1, min): max = fnmax(x1, max)
                                END IF
                        NEXT
                        IF min + max > 0 AND min < 1 AND max < 1 THEN
                                dx = ABS(min - max) * 2 / (min + max)
                                leq = fnsatisfy(.01, .2, dx)
```

```
                ELSE
                        leq = 0
                END IF
                leqmin = fnmin(leqmin, leq)
        NEXT
        lamda = fnmin(lamda, leqmin)
        f = 1 - lamda
    ELSE
        f = 101010
    END IF
END SUB
```

4.3.3 CHANGING TO THE CONJUNCTION OPERATOR FOR λ_X : REPLACEMENT FOR SUB SATISFY

```
SUB satisfy (s%, c%, user(), regl(), maxr, f(), lamda)
SHARED t%, z%(), lamdax, lamdar
min = regl(1): max = regl(2)
lamdar = fnsatisfy(min, max, maxr)
'---calculate user satisfaction---
lmdx = 0
FOR i% = 1 TO c%
    lamdai = 100: lamdax = 100
    FOR j% = 1 TO s%
        IF (j% = 1 AND z%(i%) = 1) OR (j% > 1 AND j% < s% AND z%(i%) = 0) OR j%
= s% THEN
                min = user(j%, i%, 1): max = user(j%, i%, 2)
                v = f(j%, i%)
                lx = fnsatisfy(min, max, v)
                lamdai = fnmin(lx, lamdai)
        END IF
    NEXT
    lamdax = fnmin(lamdax, lamdai)
NEXT
lamda = fnmin(lamdar, lamdax)
END SUB
```

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RISK AS A TOOL IN WATER RESOURCE MANAGEMENT.

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Voorgelê ter vervulling van 'n deel van die vereistes vir die graad *Philosophiae Doctor* in die *Fakulteit Ingenieurswese, Bou-omgewing en Inligtingstegnologie*, Universiteit van Pretoria

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Title: **Risk as a tool in water resource management**

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Synopsis

The National Water Act (Act 36 of 1998) (NWA) of South Africa makes provision for a quantity and quality of water to be set aside as a Reserve for the provision of basic human needs and for the protection of the aquatic ecosystem for sustainable development of the water resource. An ecological risk approach to water management with a view to the Reserve based *inter alia* on the following:

- Ecological risk is explicitly effect oriented.
- A risk approach will not only address the stochastic characteristic of the ecosystem, but it will also provide a useful tool to address the potential conflict between user and legislator. A risk approach is explicitly effect oriented.
- The probability component of risk supplies a way to bring diverse stressors to a common basis and address the diverse-stressor-multiple source problem.

This study aims to provide a tool to apportion the ecological effect impact attenuation rationally among users.

In order to accomplish this, attention was given to the following:

1. The end-point required by the NWA must be related to end-points at lower organisational levels of the ecosystem. A model is proposed to do this based on the logical relationship between ecological phenomena. Although there is a dearth of information to use in the model, it may contribute to the characterisation of uncertainty with this type of projection.

2. The mathematical formulation of the ERA process has apparently not received much attention in the technical literature. A mathematical formulation of the risk of a single stressor is proposed in both probability and fuzzy logic terms. The risk is expressed as the conjunction of the likelihood of effect conditioned on the stressor occurrence and a likelihood of stressor occurrence.

3. When diverse stressors occur together and no other information is available on their interactions, the aggregate stressor risk may be expressed as the disjunction of individual stressor risks. The value of this approach is investigated in some hypothetical but realistic case studies.

4. The problem of apportionment of impact attenuation burden among multiple dischargers of diverse stressors is similar to waste-load allocation (WLA). Obtaining an equitable distribution of the effect attenuation burden that recognises the technological and economic limitations in a catchment, is an optimisation problem. The diverse-stressor-multiple-source problem is first formulated as a fuzzy optimisation problem, which is solved using a genetic algorithm. This approach is investigated in a hypothetical (but possibly realistic) case study. The objective of the optimisation is the maximisation of the acceptability of the regulated situation. For the regulator

this is assumed to mean the minimisation of ecological risk, while for the stressor source manager this might be influenced by technological and economic considerations. The degree of attenuation of the stressor is chosen as the control variable.

Key terms: Ecological risk; Probabilistic risk; water quality management; fuzzy logic; fuzzy risk; optimisation; Water Act.; Resource management.

Samevatting

Die Nasionale Waterwet (Wet 36 van 1998) (NWW) bepaal dat 'n bepaalde hoeveelheid en gehalte water opsy gesit word as 'n Reserwe vir basiese menslike gebruik sowel as vir die beskerming van die akwatiese ekostelsel. Daarbenewens, word die verpligting op die staat geplaas om die waterhulpbron volhoubaar te ontwikkel. Die ontginning van die hulpbron sal kennelik druk plaas op die akwatiese ekostelsel. 'n Ekologiese risiko benadering in hulpbronbestuur word voorgestel, ondermeer omdat:

- Ekologiese risiko is eksplisiet effek georiënteerd.
- 'n Risko benadering tot hulpbronbestuur sal nie net die stogastisiteit en onsekerheid wat die ekostelsel kenmerk, kan aanspreek nie, maar voorsien ook 'n veelsydige stuk gereedskap wat gebruik kan word om die potensiële konflik tussen gebruiker en beskermer aan te spreek.
- Die waarskynlikheidskomponent van risiko bied 'n manier om diverse stressors op 'n gemeenskaplike basis te plaas om die diverse-stressor-veelvuldige-bron probleem aan te spreek, d.w.s. dié probleem waar diverse stressors wat in verskillende eenhede uitgedruk word maar tot dieselfde globale effek bydra en daarbenewens nog uit verskillende bronne kom, te bestuur.

Hierdie studie poog om die gereedskap te ontwikkel wat die ekologiese impakbekampingslas op 'n rasionele basis tussen gebruikers toe deel.

Ten einde hierdie doel te bereik word aandag gegee aan die volgende aspekte:

1. Die eindpunt (tw. volhoubaarheid) wat deur die NWW vereis word moet in verband gebring word met eindpunte by laer organisasie vlakke van die ekostelsel. Hiervoor word 'n model voorgestel wat gebaseer is op die logiese verband tussen ekologiese verskynsels. Hoewel besonderhede vir die model skaars is, kan dit bydra tot die uitspel van onsekerheid by hierdie vorm van eindpunt projeksie.

2. Die wiskundige formulering van ERA het min aandag in die vakliteratuur gekry. 'n Wiskundige uitdrukking van risiko skatting vir 'n enkele stressor word voorgestel in beide waarskynlikheidsleer formulering en newellogika (Eng. "fuzzy logic") formulering. Die risiko vir 'n stressor word uitgedruk as die konjunktiewe samestelling van die verwagting van effek gekondisioneer op die stressor voorkoms en die verwagting van die stressor voorkoms.

3. Wanneer diverse stressors saam voorkom, en geen verdere inligting beskikbaar is oor hulle wisselwerking nie, word die gesamentlike risiko voorgestel as die konjunktiewe samestelling van die afsonderlike risiko's. Die waarde van hierdie benadering word getoon aan die hand van hipotetiese maar realistiese gevalle studies.

4. Die probleem van toebedeling van impakbekampingslas tussen veelvuldige stressorbronne is soortgelyk aan die afval-beladingstoebedeling ("waste load allocation") probleem. Om 'n

eweredige effekbekampingslas te verkry wat die ekonomiese en tegnologiese beperkings van verkillende watergebruikers in die opvangebied in aanmerking neem, is 'n optimiseringsprobleem. Die diverse-stressor-veelvuldige-bron probleem word eers as 'n newel optimiseringsprobleem geformuleer wat dan met behulp van 'n genetiese algoritme opgelos word. Die benadering word aan die hand van 'n hipotetiese (maar moontlik realistiese) gevallestudie ondersoek. Die doelwit van die optimisering is die maksimisering van die aanvaarbaarheid van die gereguleerde situasie. Vir die wetstoepasser is die beperkings van ekologiese risiko waarskynlik belangrik terwyl koste en tegnologiese faktore waarskynlik vir die stressor bestuurder belangrik is. Die graad van stressor vermindering is as beheerveranderlike gekies.

Sleutelterme: Ekologiese risiko; waarskynlikheidsrisiko; watergehaltebestuur; newellogika; newelrisiko; optimisering; waterwet.; hulpbronbestuur

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Abbreviations

Λ or L:	A likelihood measure (such as probability, possibility or necessity)
AEL:	Acceptable effect level
AEV:	Acute effect value (from the SAWQG)
ASL:	Acceptable stressor level
BCF:	Bio concentration factor
BEL:	Benchmark effect level
CAP:	Continuous assessment paradigm (see Appendix Chapter 1)
DO:	Dissolved oxygen
DSMS:	Diverse-stressor multiple-source
EQO:	Environmental quality objective
ERA:	Ecological risk assessment
ERBM:	Ecological risk-based management
ESL:	Expected stressor level
GA:	Genetic algorithm (for optimisation)
Inf:	Infimum (lowest lower bound)
LBB:	Lethal body burden
LC50:	Median lethal concentration
Max:	Maximum
Min:	Minimum
MOA:	Mode of action
MOOP:	Multiple objective optimisation problem
NOEC:	No observed/observable effect concentration
NWA:	National Water Act (Act 36 of 1998)
QAP:	Quantal assessment paradigm (see Appendix Chapter 1)
RDM:	Resource directed measure (provided for in the National Water Act)
RO:	Risk objective
SAWQG:	South African Water Quality Guidelines (1996 edition)
SDC:	Source directed control (provided for in the National Water Act)
SRR:	Stressor response relationship
Sup:	Supremum (highest upper bound)
WET:	Whole effluent toxicity
WLA:	Waste-load allocation

Definitions

{} denotes a set of discreet values, [ ] denotes a continuous interval, *sup*{...} is the highest upper boundary of the set, and *inf*{...} denotes the lowest lower boundary of the set.

Biodiversity: "The variety of life at all levels of organization, represented by the number and relative frequency of items (genes, organisms and ecosystems)"(USEPA, 1997a).

Degree of membership (μ): The Zadehian view: The degree of membership of a value x to fuzzy set A $\mu_A(x)$ is a function which describes the congruence of the perception of x the qualification(s) of A (it expresses the "A-ness of x"). This view supposes that the datum is vague and therefore that μ is the extent to which an observation agrees with the vague concept. The epistemic view (Kruse, *et al.*, 1994): μ is a probability distribution of how well an observation coincides with a specific datum which is only known with uncertainty. It differs from probability in that (*inter alia*) while probabilities sum to 1, in general, membership functions do not.

Ecological risk assessment (ERA): the technique that "evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors"(EPA, 1996). In practice it is the application of the science of ecotoxicology to public policy (Suter, 1993).

***Epistemic*:** Dealing with the nature of knowledge and understanding.

Fuzzy logic: A branch of logic that deals with an infinite number of truth values. If x represents the truth value of a statement, then in Boolean logic $x \in \{0,1\}$ while in fuzzy logic $x \in [0,1]$.

***Hazard*:** The potential of a substance or situation to cause harm.

Integrity: "The state of being unimpaired, sound" (DeLeo and Levin, 1997), "the quality or condition of being whole, complete". The functional definitions are more diverse: "the interaction of the physical, chemical and biological elements of an ecosystem in a manner that ensures the long term health and sustainability of the ecosystem" (USEPA, 1997a), or "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a full range of elements (genes, species and assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat of a region" (Karr, 1996). Other definitions appear to be subsets of these definitions (Cairns, 1977, Karr and Dudley, 1981, Noss, 1990, Rapport *et al.*, 1996).

***Likelihood*:** An expression of the sense of expectation of an observer about an event whether based on repeated observation of identical or morphologically similar events. Can be expressed in terms of probability or possibility (fuzzy) theoretical terms.

Necessity measure: The necessity measure $\mathrm{Nec}_\pi(\mathrm{A}) = inf\{1 - \pi(\omega) \mid \omega \in \Omega \backslash \mathrm{A}\} \in [0,1]$. The necessity measure is related to the possibility that the uncertain event $\omega 0$ belongs to the universal set Ω without the set A and is therefore a stonger measure indicating that $\omega \in$ A than the possibility measure.

P(A /B) **:** The probability of A conditional on B.

***P(AB) or P(A$\wedge$B)*:** The probability of A and B; or the probability of A in conjunction with B.

***Phenomenon*:** That which appears real to the senses regardless of whether the underlying existence is proved or its nature understood.

Possibility measure *:* A measure of the possibility that an event may occur. The possibility measure for event A, $\Pi_\pi(A) = sup\{\pi(\omega) \mid \omega \in A\} \in [0,1]$. If the possibility of an event is 1 it is entirely possible, while 0 indicates that the event is not possible. The possibility measure does not give any indication of the probability of an event.

Resilience: "The ability of an ecosystem to adapt to change (or stress)" (USEPA, 1997a), or, "the ability to maintain integrity when subject to disturbance" (Holling 1973).

Risk**:** "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1993) or the probability of observing a specified (unacceptable) effect as a result of a toxic chemical exposure (Bartell, *et al*, 1992). In essence, whether explicitly or implicitly, risk contains elements of: a) likelihood, b) target and c) unacceptable effect. The manner in which the likelihood is expressed introduces gradations to the concept: when a situation allows for Aristotelian (binary) logic and likelihood can be expressed as a probability, then the common form of risk assessment is recovered. However, when fuzzy logic is required and likelihood is expressed in possibilistic terms then fuzzy risk assessment is called for.

Sustainability **:** "the ability of an ecosystem to support itself despite continued harvest, removal, or loss of some sort" (USEPA, 1997a). Implicit in this definition is the assumption that sustainability is time and stressor dependent.

t-norm and t-conorm: Used to define generalised intersection and union operators respectively for fuzzy sets.

Truth value: The truth value of a proposition is the degree to which the content of the proposition agrees with the assessors perception of reality. The truth value can be calculated as the compatibility of the possibility distribution representing the proposition with the possibility distribution representing the state of knowledge (Du Bois and Prade, 1988, p126)

Structure

This document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

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Part 1:

Overview

PART 1: OVERVIEW

1. BACKGROUND

This study originated from the thinking around the South African National Water Act (NWA) (Act 36 of 1998) which replaced an older Act dating from 1956. Three aspects of the NWA had a particular impact on this study:

The NWA guarantees only two rights: sufficient quantity and quality of water to supply basic human needs and to ensure the sustainable functioning the aquatic ecosystem. This quantity and quality constitutes a Reserve, which needs to be protected.

It makes provision for measures to protect the resource as well as to control sources of pollutants (or stressors).

It makes provision for a classification system for resources.

> An ecological **stressor** could be any **substance**, group of substances, a **flow-related** quantity, an in-stream- or riparian **habitat condition** or presence of biota that is **not normally expected** at a given time and place

This study deals particularly with the ecological requirements; briefly referred to as the "ecological reserve". (For more detail on the NWA and its requirements see Part 2, Chapter 1.)

The concept of an ecological reserve developed from the notion that ecosystems are generally fairly resilient and if they are not "pushed too far", they can usually regain the level of services practically indistinguishable from the pre-impact level. It was reasoned, however, that there may be a point at which the system is "pushed too far" so that it then "crashes". A "crashed" system would of course be undesirable, but exactly what constitutes that "crash-point" is uncertain. All that seems reasonable to assert is that the more the system is "pushed" (in the sense of moved away from pristine condition), the greater the likelihood the system will "crash". So, in a broad and as yet undefined sense, the further the system moves from its pristine state the higher the risk of system "crash". From these vague roots the concept of "risk" and particularly "ecological risk" intuitively appeared to be useful. The resulting "grey scale" of risk can be discretised to serve as the basis for a classification system for resources where one end of the scale represent insignificant risk while the other represents unacceptable risk.

This study proposes the use of ecological risk as a decision support tool in water resource management in support of the protection of the ecological reserve. "Ecological risk" and "ecological risk assessment" have become fairly well established as a decision support tool in environmental management as is shown by the literature cited in Parts 2 and 3. The terms "risk" and "risk assessment" have come to take on a wide variety of meanings and encompass a wide variety of practices. This study attempts to find a suitable expression of risk and examines some theoretical concepts around its application to water resource management.

> **Ecological risk assessment (ERA) for the aquatic environment under the NWA should estimate the likelihood that loss of sustainability will result from the occurrence of aquatic stressors**

This study lays no claim to providing new insights into ecological mechanisms that are involved in vague terms like "system crash", "pushed too far". It accepted that there are experts in biology and ecology who can produce elegant, precise and scholarly definitions for these vague terms. As a point of departure, these are used in a phenomenological sense, i.e. without knowing the biological and ecological mechanisms, "pushed" simply refers to the phenomenon "inducing a movement away from" and "crashed" simple refers to a phenomenon "not being able to produce what is expected". So, where some more precise terminology is used, it must be accepted that these are from a relative layman's point of view. It is hoped that where more precise information becomes available, it will still be useful within the theoretical framework provided here with some adaptation of the methodology.

2. GOALS

In this study three main issues are addressed:

1. The rationale for the use of ecological risk - Is risk really conceptually useful in water resource management with the aim to ensure sustainability?
2. Is there a mathematical construct that could be used for risk calculation in ecological risk assessment in the NWA context?
3. How could risk be applied in a multiple stressor multiple source environment?

3. RATIONALE FOR THE USE OF ECOLOGICAL RISK

"No, no!", said the Queen. "Sentence first - verdict afterwards"
– Lewis Carroll, Alice in Wonderland

Decisions regarding water quantity and quality often have to be made based on meagre information, the impact of which may either justify or condemn the decision.

The unenviable task of the water resource manager may at times seems to call for the reasoning of the queen during the trial in *Alice in Wonderland.* Decisions regarding water quantity and quality often have to be made based on meagre information, the impact of which may either justify or condemn the decision. The reason for this is rooted both in the characteristics of the aquatic ecosystem and our knowledge and use of it. This section addresses the first goal of the study.

3.1 SOME FUNDAMENTAL ISSUES

The event referred to as "ecosystem crash" is a manifestation of impact on the specific assemblage of aquatic organisms making up that ecosystem. The identity of the organisms, their interactions and their relative abundances are determined by a number of both biotic and abiotic factors. In the pristine state, these factors are in dynamic equilibrium, identifying the reference condition for describing system integrity. Now three very fundamental assumptions have to be made:

- Pristine, un-impacted ecosystems do not "crash". Even extreme hydrological events such as floods or droughts are part of the natural regime of ecosystems.
- Aquatic organisms would react to a change in the natural state of their physical, chemical, and biological environment.
- This "crash" only takes place when an unnatural condition is imposed on the system, such as by anthropogenic intervention. Deviation from the pristine state of the ecosystem (interpreted as loss of biotic integrity) would increase the likelihood of reaching that "crash point". The pristine state defines the condition of trivial (or *de minimis*) risk while the crash point defines a condition of unacceptable (or *de manifestis*) risk.

> Extreme natural events such as droughts and floods, which are part of the pristine state regime, are not considered as stressors.

So, in principle sustainable ecological water resource management is simple: manage the physical, chemical and biological environment within suitable limits and system "crash" will be avoided. But what are those "suitable limits" providing a suitable margin of safety?

3.2 COMPLICATING FACTORS IN ECOLOGICAL RESERVE MANAGEMENT

Determining the suitable limits for management is complicated by noting that in dealing with the ecological reserve, or any system where ecological sustainability is an issue, scientists and managers have to address:

Vaguely defined systems (see Part 2 Section 2.3.2 and Part 3 Paper 1)
When dealing with the impact of some form of water use on a specific river reach it could be argued on the one hand that the entire globe is one big ecosystem with internal links of different strengths. On the other hand it could be argued that only the individual organisms in that reach and their direct interactions constitute the ecosystem. To a certain extent both are correct. Between these two extremes system boundaries are a matter of opinion. Of course, in each river or stream and in any given reach of that stream the identity of organisms that make up the system would be different, their individual susceptibilities to environmental factors would be different, and their interactions would be different.

Fragmentary knowledge and uncertainty in its interpretation (see Paper 1, Part 3).

While extensive systematic studies have been performed on certain aquatic species, knowledge of the interaction among species and between species and their environment is not always as well developed. While toxicology (the science of the interaction of substances and individual organisms) has developed into a reasonably exact science, the same cannot always be said for ecotoxicology (the science of the interaction between substances and ecosystems). Even where extensive observations of stimuli and their responses are available, the interpretation of the results is not always uniform. Different conceptual approaches to looking at the same set of observations leads to different models of the system under observation. Different models may yield different assessments of future system response. Different assessments may, in turn, lead to different management strategies.

Systems that are subject to various forms of randomness (see Section 2.3 in Part 2and Papers 1 and 2, Part 3).

In contrast to the previous problem that could conceivably be resolved by more intensive study, randomness is not reduced by study. Randomness (or stochasticity) is often an integral part of ecosystem dynamics. Randomness in ecosystem response is also influenced by randomness in the hydrological cycle (e.g. rainfall, run-off etc.) and by individual variability in response to stressors. The problem, of course, usually arises when the mind-set is deterministic.

A variety of different stressors, each of which may to a greater or lesser extent have an impact on the aquatic ecosystem (see Part 2, Chapter 3 and Part 3, Paper 1).

Conventionally, undesirable substances or energy (in the form of heat) added to water were considered important. However, the amount and timing of water supply and in-stream and riparian habitat condition are also important and may, in some cases, even be more important than water quality in determining ecological impact. Each of these is quantified in different units. Each of these may cause "ecosystem crash". How does one decide on the seriousness of the combined impact? In order to facilitate management, it would be useful (if not necessary) to rank these stressors on a common basis.

Ensuring environmental protection while at the same time not stifling progress (see Part 2, Chapter 4 and Part 3, Paper 4).

Theoretically it is simple to take a precautionary approach when dealing with multiple stressors – to select levels of these stressors where there would be no known effect. However, in a developing, water scarce country like South Africa, this is not so easy. There is a significant need for economic upliftment and development in what is otherwise a frail economy. Water treatment facilities range from highly sophisticated to non-existent. In large areas of the country agriculture is dependent on irrigation from surface water resources and dilution capacity is very limited. An entirely precautionary approach in water resource management may, in some areas, have a devastating economic and sociological effect.

All of the above contribute to an unenviable management situation. From the above, it would appear to be practically impossible to define which set (or sets) of values of physical, chemical and biological variables define that "crash point" and without that information it would impossible to define what a safe margin would be. All that can reasonably be assumed is that the likelihood or probability of ecosystem "crash" increases as deviation from pristine levels increases.

3.3 APPRAISAL OF RISK AS RESOURCE MANAGEMENT TOOL

Some of the important and useful characteristics of risk include:

a. Risk makes use of **two important types of information**: What we know about what would happen to a system when it is exposed to a stressor (i.e. an **effect assessment**), and what we know about the stressor's occurrence (i.e. an **occurrence assessment**). The first question is the basis for a hazard assessment. It does not concern itself with how the stressor behaves in the real world. What risk as a methodology does is to bring the stressor occurrence characteristics in as part of the assessment.
b. Ecological risk needs an **end-point**, i.e. a specific expression of what sort of effect is being assessed. In the case of the ecological reserve, the end-point required by the NWA is "loss of sustainability" (that is the "statutory" end-point). This end-point has a specific value for the public. On the other hand, the scientists who have to assess the impact of a stressor usually don't really have any information specifically relating to "loss of sustainability" as such, but they may infer "loss of sustainability" from other information such as "disappearance of a key species" (that is a "surrogate end-point"). Both statutory and surrogate end-points may be subject of debate and/or negotiation. Projecting from the surrogate to the statutory end-point is not trivial (see Part 2, Chapter2 and its Appendix and Part 3, Appendix to Paper1)
c. A particular characteristic of risk (in the technical sense used here) is its **expression in terms of likelihood** (e.g. probability). If the end-points for the assessment of risk resulting from different types of stressors are the same, then likelihood is practically a unitless way of comparing and **expressing the impact of diverse stressors** (see Part 2 Chapter 3 and Part 3, Papers 2 and 3). This is because the likelihood expression is equipped to handle the complicating factors above better than a hazard approach.

Dealing with technical issues in resource management for the protection of the ecological reserve		
Issue	**How issue can be addressed on a risk basis**	**Further Information**
Uncertainty in models and innate randomness (stochasticity)	Calculation of probabilistic risk. Can be expressed as uncertainty in the calculated risk	Part 2, Chapter 3 and Part 3, Paper 2
Vaguely defined systems and fragmentary knowledge	Possibilistic risk based on fuzzy logic	Part 2, Chapter3 and Part 3, Paper 3
Assessing impact for a diversity in stressors	Risk aggregation	Part 2, Chapter 3 and Part 3, Papers 2 and 3.
Relating the regulatory (statutory) end-point for an assessment the surrogate end-point	Projection model for assessment confidence	Part 2 Chapter2 and Example in Part 3, Paper 1.
Deriving criteria for the management of multiple sources of diverse stressors	Optimisation to risk objectives	Part 2, Chapter 4 and Part 3, Paper 4

d. A risk approach tends to be **less wasteful of available information** than a hazard approach to stressor management. As indicated in a), a hazard approach tends toward focussing on critical effect benchmark values, i.e. stressor levels that represent selected levels of effect that are perceived to be important by role players in the assessment process. How effect-levels change at stressor levels above and below the benchmark is neglected in the assessment. The major effort in a hazard assessment is focussed on how the stressor presents itself. A risk approach has the potential (even if not always used as such) of being able to utilise both types of information. (See Part 2, Appendix 1 for a discussion of the risk and hazard paradigms). In addition, it is a vehicle to expresses some forms of uncertainty and its impact on a situation assessment (see Part 3, Paper 2).

Because of the factors above risk is also a more arduous approach to resource management. The extra effort pays off by providing a very versatile decision support tool. It is possible, for example, to trade off stressors against each other once a risk goal for a resource has been set. This is particularly useful in addressing factor 5 above (the diverse stressor multiple-source problem, see Part 2 Chapter 4 and Part 3 Paper 4).

The likelihood component of risk can be expressed either qualitatively or quantitatively. Expressions of likelihood can be based either on probability theory, which has a strong mathematical and historical underpinning, or it can be based on fuzzy logic, which has an advantage in dealing with vague expressions often encountered in descriptive ecology. The most suitable expression will depend on the application.

3.4 RISK OBJECTIVES

In applying risk in a resource management framework two types of application can be distinguished: using risk merely as a ranking tool, where the actual risk magnitudes do not matter, or, using risk explicitly.

In the latter case it is assumed that risk objectives will be generated. Risk objectives (e.g. the probability of the loss of species should be $< 10^{-4}$) would be analogous to other forms of in-stream objectives, with the exception that they are essentially dimensionless (referring only to an undesired effect, such as loss of sustainability).

4. TECHNICAL ISSUES IN USING RISK

In addressing the complicating factors in resource management in support of the ecological reserve (above) a number of technical issues needed to be addressed.

4.1 DEFINITION OF RISK

A variety of definitions for risk were encountered in environmental risk assessment literature. For the purpose of this study risk was defined as the likelihood that a loss of sustainable ecological function will occur (Part 2, Paper 1).

4.2 ESTIMATION OF RISK

From the discussion of the components of a risk assessment (Part 3, Papers 1 and 2) a risk assessment should combine a likelihood assessment of effect with a likelihood assessment of occurrence. A number of methods were encountered:

Ratio of benchmarks

The Predicted Environmental Concentration to (Predicted) No-Effect Concentration ratio is one example. If the ratio is less than 1 then no risk exists while if larger that 1 a risk exists. This appears to be little more than a hazard assessment in weak disguise.

Probability of effect benchmark

This requires the calculation of the probability that the environmental concentration will be larger than a benchmark concentration. This still does not provide information on what would happen if the concentration is larger than the benchmark concentration.

Degree of overlap

This method involves determining the area of overlap between an effect likelihood curve (expressed as the likelihood of effect vs. stressor level) and the stressor occurrence likelihood curve (like the probability density function of stressor level occurrence). While conceptually simple, it is not quite clear how to interpret the result.

> The event conjunction model is useful for calculating a stressor-specific instantaneous risk. The stressor-specific risk may be calculated from either the maximum instantaneous risk or from the cumulative risk for a specific situation.
>
> The aggregate risk could be estimated from the disjunction of stressor-specific risk.

Occurrence and effect event conjunction

In general the risk assessment literature recognises that risk depends on some form of conditional probability. As far as could be established, this type of formulation does not appear in the ecological risk assessment literature referenced in this study.

From a theoretical perspective it seemed feasible to assert that a risk only exists when two events occur simultaneously: the event that a hazard exists and the event that a stressor occurs. As a corollary to that one might say that a stressor is only defined as such when it can result in the undesired effect that is chosen as the end-point (see Part 2, Chapter 3 and Part 3, Papers 2 and 3). Consequently, risk was defined as the likelihood that a specific level of effect will occur conditioned in the occurrence of a specific stressor level, in conjunction with the likelihood that this specific stressor level will occur (see Part 2, paper 2 and Part 3, 3.3).

So if E is the undesired effect and x is a level of stressor X, then the risk $R_X = L(E|x)*L(x)$, where L is a likelihood operator such as probability, possibility or necessity and * is a corresponding conjunction operator such as multiplication in the case of probability or maximum or minimum in the case of possibility and necessity.

R provides an estimate of the risk pertaining to that specific level of stressor ("instantaneous risk"). In order to assess the risk pertaining to a situation where a spectrum of stressor levels are possible, two approaches can be taken:

- The cumulative distribution of the instantaneous risk can be determined (this approach was used in Part 2, Chapter 3 and Part 3, Paper 2), or
- The maximum value of the instantaneous risk over all possible stressor levels can be determined, i.e. the likelihood that the system will experience the undesired effect can be no higher than the most likely instantaneous event. This is the basis of the fuzzy approach (Part 2, 3.4 and Part 3, Paper 3).

The Kelly-Roy-Harrison expression

Subsequent to submitting the papers in Part 3 the paper by Kelly and Roy-Harrison (1998) was discovered that gives a mathematical construct of ecological risk. This expression is meant to assess different consequences of a given stressor occurrence. If the consequences are discounted in one single end-point, it can be shown that this expression is a special case of the general inference scheme on which the above formulation is based (Part 2, Chapter3, 3.2)

4.3 END-POINT PROJECTION

One of strengths of the ecological risk approach is the requirement to establish clear end-points. This contributes to making the assessment transparent. As pointed out in Section 3.3 b) above, the statutory and surrogate end-points often do not coincide. An end-point projection model needs to set be up. An example of such a model is given in Part 2, Section 2.4.3 and Appendix 2, Sections 2.10.1 to 2.10.4 and Part 3, Paper1). This model is meant as a prototype to indicate what sort of inputs might be necessary and (qualitatively) how this might influence confidence in a risk assessment.

4.4 APPLYING RISK TO THE DIVERSE STRESSOR MULTIPLE SOURCE PROBLEM

A generalised scheme for the application of risk methodology in resource management and particularly with respect to establishing desired resource management stressor criteria, is shown in the figure below (see Part 2, Section 2.2.3)

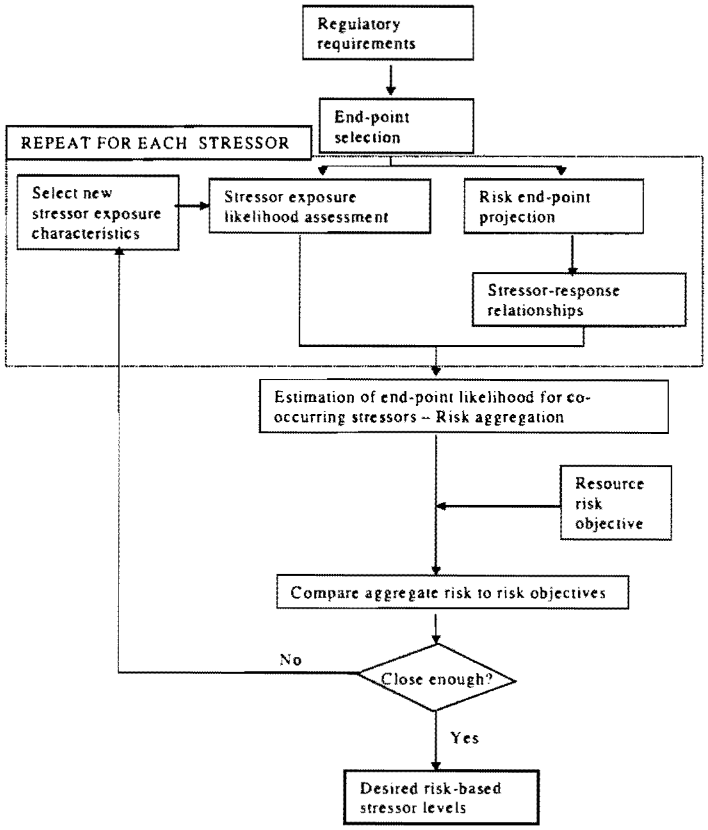

Diagram of a generic application of ecological risk-based management showing how aspects of the ERA process could be used. Detailed discussion appears in Part 2 Chapter 2.

4.5 AGGREGATE RISK

An important advantage in a likelihood expression of risk is the ability to compare stressors directly. The implication here is that identical end-points are used in the stressor specific risk assessment. Furthermore, stressor risk can be assumed to be logically independent, i.e. the occurrence of an effect due to one stressor does not imply the same effect due to any other stressor. (Logical dependence needs to be distinguished from mechanistic dependence where effects such as additivity, supra-additivity or infra-additivity might be at work and which will influence conditional effect dependence in the instantaneous risk assessment).

With this being the case, simple probability and possibility theory suggests modelling the aggregate risk as the disjunction (or union in set theoretical terms) of logically independent events. Examples are provided in Part 2 Section 2.5 and 2.6 and in Part 3, Papers 2 and 3 for probabilistic and fuzzy risk respectively.

4.6 APPLYING A RISK OBJECTIVE: THE DIVERSE-STRESSOR-MUTIPLE-SOURCE PROBLEM

Up to this point only a typical risk assessment scenario has been addressed where a situation exists where a stressor or stressors occur or may occur and the goal is to assess the resulting risk. However, the situation is somewhat more complex when one has to manage stressor levels to an ecological risk goal (Ecological risk-based management, ERBM).

This is analogous to waste-load allocation where an in-stream water quality objective is given and it is necessary to derive point source criteria to meet an in-stream objective. The problem now is that many different combinations of stressor-levels result in same risk. Therefore, additional information is required to decide on suitable source criteria. This apparent obstacle can be turned into advantage since it provides the opportunity to incorporate independent information (independent with respect to biological effect or exposure) into the assessment. Optimisation is required to solve this problem (see Part 2 Chapter 4 and Part 3 Paper 4).

The fuzzy optimisation problem was formulated as finding that set of stressor source attenuation values that maximised the overall acceptability of the regulated situation. It was assumed that the regulator would be satisfied when the risk was minimised but with a maximum threshold. On the other hand, the regulatees would be satisfied with minimised stressor attenuation with a graded acceptability between completely unacceptable and completely acceptable. Various ways of estimating the overall satisfaction were investigated, each relating to policy decision by the regulator.

Both Simplex and Genetic optimisation algorithms were explored but the genetic algorithm was found to be the most suitable.

5. GENERAL CONCLUSIONS

See also Part 2, Chapter 5 for more detailed discussion.

Is risk really conceptually useful in water resource management with the aim to ensure sustainability?

Ecological risk, formally defined as the likelihood that loss of sustainability will occur, is potentially very useful in the context of the NWA. In principle it addresses most of the major factors impacting on the uncertainty in ecological assessments at least semi-quantitatively. It could:

- Serve as a rational basis for classifying resources where the classification would take into consideration both what is known about the stressor effect on the system and what is known about the stressor's actual likelihood of occurrence.
- Be used in the management of highly utilised catchments as a tool to formulate policy and derive source and stressor specific management criteria.

Is there a mathematical construct that could be used for risk calculation in ecological risk assessment in the NWA context?

A theoretically sound way of assessing risk is presented in this study. It comprises a conjunctive stressor–specific risk estimation and a disjunctive risk-aggregation. This mathematical formulation is extended both to the probabilistic and possibilistic domains. It is computationally easy and it can be coded for spreadsheet use for resource classification purposes.

How could risk be applied in a multiple stressor multiple source environment?

a. **Ranking stressors** is simple enough on a risk basis.

b. Risk has the potential to be used as the basis for **stressor specific resource quality criteria**. The advantage would be that all stressors would then be comparable on the basis of the same effect. This aspect needs further development.

c. **Classification of resources** with a view to setting the reserve. In order to accomplish this it would be required to set ecological risk goals for resources and/ or classes of resources. This aspect needs further development.

d. Deriving source- and stressor-specific management criteria in catchments with high pressure for resource use. This would require co-operative effort from water users who have to be able to formulate ranges within which they are able to attenuate the stressors they produce. Computationally this is quite demanding but in cases where there is economic pressure this may pay off handsomely both to the regulator and the regulatees.

Two issue merit critical attention:

Deriving stressor-response relationships. Risk characterisation/ calculation remains critically dependent on the quality of the knowledge of the relationship between stressor occurrence and the corresponding response. In this study that knowledge was modelled either as a stressor-response relationship (that describes the likelihood of observing an end-point as a function of stressor level) or as a rule base formulating the same type of knowledge on a more qualitative basis. Methodology is needed to formalise the derivation of these relationships from experimental observation and/or expert opinion.

Deriving/ setting ecological risk objectives for streams. The success of risk-based management is critically dependent on acceptable risk objectives. Two aspects in particular need attention: acceptability to the water use community and acceptability to the scientific community.

Structure

The document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: (This Part) Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

Part 3: Presents some of the papers that have been published in peer reviewed literature and that are included for quick reference.

Part 2:

Technical discussion

CHAPTER 1

INTRODUCTION AND BACKGROUND

1.1 SUMMARY

In the South African context, the National Water Act supplies the regulatory background for water resource management. The provision of a suitable quantity and quality of water for basic human needs and sustainable use of the aquatic ecosystem as a Reserve, supplies the regulatory background for water resource management. This has to be balanced with the development needs within the water use community. The uncertainty and variability inherently part of the ecological knowledge base, which complicates this process, can be addressed by ecological risk expression. This supplies the basis for a continuous assessment of effect, which is necessary to find the optimal state between the satisfaction of ecological goals on the one hand, and the operational requirement for managing the system on the other hand. Specifically this study addresses:1) The systematic basis for deriving ecosystem level end-points from stressor occurrences, 2) Expressions of ecological effect likelihood and their convolution as a basis for the expression of overall effect expectation, 3) The optimisation procedure for estimating stressor attenuation levels in order to achieve ecological goals, and 4) An application framework for this derivation procedure.

1.2 INTRODUCTION

The South African National Water Act (Act 36 of 1998) (NWA) makes provision for the protection of a Reserve. The Reserve refers to a quantity and quality of water that will assure the supply of water for basic human needs as well as the sustainable functioning of the aquatic ecosystem (DWAF, 1997). The NWA contributes by giving effect to the right to a healthy environment as guaranteed by the South African Bill of Rights. In fact, the protection of the Reserve is the only right with regard to water under this Act. The NWA also does away with the *dominus fluminis* principle of the Roman Dutch law, which gives a riparian landowner the right to use of the water in the stream. Water is viewed as a resource to which all South Africans should have reasonable access and which is administered for the common good by the state.

1.3 REGULATORY BACKGROUND

In terms of the NWA, it should be noted that:

- ✓ The term "quality" is defined so as to include not only the chemical and physico-chemical components of the water, but also the integrity of biota, the assurance of flow and the habitat structure.
- ✓ The water resource includes, not only the water column of streams and rivers, but also the ground water, sediment and estuaries as well as the riparian habitat. Consequently, when reference is made to "resource quality", it encompasses virtually all manageable aspects of practically all compartments of the water environment (except the water/air interface).
- ✓ The aim of the NWA, besides the protection of the aquatic ecosystem and the supply of basic human needs, is to prevent or reduce pollution. "Pollution" refers to any alteration of the physical, chemical or biological properties of the resource that makes it harmful or potentially harmful to humans or aquatic organisms or the quality of the resource itself. The pollutants, or agents causing pollution by the definition above, are characterised by their ability to cause some form of stress (or adverse reaction) in the resource. The term "stressor" is therefore used further in the study as synonymous with "pollutant" strictly in the sense used in the NWA. This should be distinguished from a usage of the term pollutant, which mostly has the connotation of a substance that need only have a potential to cause harm.
- ✓ Under the NWA there is also a move toward a catchment management approach, as opposed to an exclusively pollutant source directed approach in water resource protection.

Although the concept of the Reserve makes provision for both human needs and that of the aquatic environment, the focus of this study is the sustainable function of the ecosystem and more specifically the application of risk methodology in water resource management. Most if not all the principles will be applicable to the human use part of the Reserve.

1.3.1 RESOURCE-DIRECTED MEASURES AND SOURCE-DIRECTED CONTROLS

The NWA makes provision for two sets of administrative tools to accomplish the goal of sustainable development of the water resource (DWAF, 1997):

1. Resource-directed measures (RDM's), which include a resource classification system that requires the grouping of significant surface water resources (among others) into protection classes. Each class represents a similar risk of damaging the resource beyond repair and corresponds to management objectives for water quality, quantity and assurance, habitat structure and biota. RDM's explicitly recognise that some damage has already occurred in the aquatic ecosystem (for example) but its point of departure is that no further degradation be allowed.

2. Source-directed controls (SDC's), which include source reduction measures that aim to reduce or eliminate the production of pollutants which could harm the water resource. SDC's will make use of permits and standards while promoting changes in technology and land-use.

> Resource-directed measures in the context of the ecological aspect of the Reserve would focus on resource protection and supply the basis of instream management objectives. The source-directed controls supply the executive means of realising resource protection. Quality criteria would necessarily be an integral part of both resource-directed measures and source-directed controls.

1.3.2 REGULATORY IMPACT OF THE RESERVE

Section 15 of the NWA makes it mandatory that any action that follows from the Act must give effect to the RDM class and its associated water resource quality objectives while Section 18 demands that such actions must also give effect to the Reserve. Section 16 determines that the Reserve must also be set in accordance with the class.

In making regulations on water use, besides giving effect to the Reserve and the resource classification system, Section 26 requires that, inter alia, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the instream and riparian habitat. Water use regulation must take into account factors such as (Section 27. (1)):

1. The socio-economic impact of water use or curtailment of use (d)
2. The catchment management strategy applicable to the resource (e)
3. The likely effect of the water use on the resource and other users (f)
4. The class and resource quality objectives (g)
5. The investment already made and to be made by the water user (h)
6. The quality needs of the Reserve and to meet international obligations (j)

> The regulatory requirement is that the SDC's must give effect to the RDM's but both of these must give due consideration to their impacts on the ecosystem and the water users. While SDC's have to give effect to the RDM's, they could be wider in their reach than RDM's and could take into consideration technology issues.

1.3.3 THE "DEVELOPMENT VS. PROTECTION" DILEMMA

From the foregoing and an analysis of the provisions in the NWA (See Appendix to Chapter 1) it is clear that:

- ⇨ The Reserve is central to water resource management in South Africa. The Reserve is the quantity and quality of water necessary to provide for basic human needs and the protection of aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource. The reserve must be given effect, not only on a site-specific basis, but also at catchment level.
- ⇨ The aspects of water that needs to be managed are diverse, including flow-, substance-, habitat- and biodiversity-related stressors. These stressors have to be managed in a way that ensures sustainability.
- ⇨ The use of the term "sustainability" implies that pressure on the ecosystem is expected and allowed. Moreover, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the insert and riparian habitat. Water use regulation must take into account factors such as the socio-economic impact of water use or curtailment of use, the likely effect of the water use on the resource and other users, the class

and resource quality objectives and the investment already made and to be made by the water user.

It is intuitively clear that resource protection, as typified by the Reserve, may somehow have to be traded off against resource development in support of other development needs. This is by no means a new problem. A simplistic formulation of this problem is "protection" (represented by a set of standards or criteria, usually with reference to the chemical and physical characteristics of water), versus "development" (represented by some economic or social surrogate measures such as "treatment cost" or "jobs lost").

Broadly, the RDM's represent the protection requirement. The SDC's on the other hand have to deal with the reality of setting end-of-pipe criteria among others, which are important for the design and operation of effluent treatment plants, for example. These relate to the economic and technical issues, which finally have socio-economic impacts. The NWA requires that RDM's and SDC's be coherent. However, in keeping with its approach to all technical matters, the NWA does not prescribe the possible approach needed to solve the problem of aligning the Reserve, RDM's and its corresponding resource quality objectives with the SDC's (such as waste discharge regulations) needed for the practical enforcement of the law.

At present the management objectives corresponding to the ecological RDM classes are set in terms of the South African Water Quality Guidelines (SAWQG, 1996; MacKay, 1999). The use of these substance/ stressor specific guideline criteria must be seen against the background of two issues: 1) The management context and 2) The diverse-stressor-multiple-source problem.

1.4 MANAGEMENT CONTEXT

Two aspects of the management in the context of the ecological Reserve are described: 1) The factors impacting on objectives and criteria in resource management and 2) Basis for formulating objectives and criteria.

1.4.1 FACTORS IMPACTING ON OBJECTIVES AND CRITERIA

The goals set by the NWA need to be translated into objectives. The objectives are the achievable "milestones" in attaining the goal. The objectives need to be translated into criteria, which are practical management values giving effect to the objectives.

The NWA goal "protection of ecological sustainability" might, with a number of assumptions, be translated to the objective "protect 95% of the aquatic species most of the time". This objective would give rise to the criteria as given in SAWQG (1996).

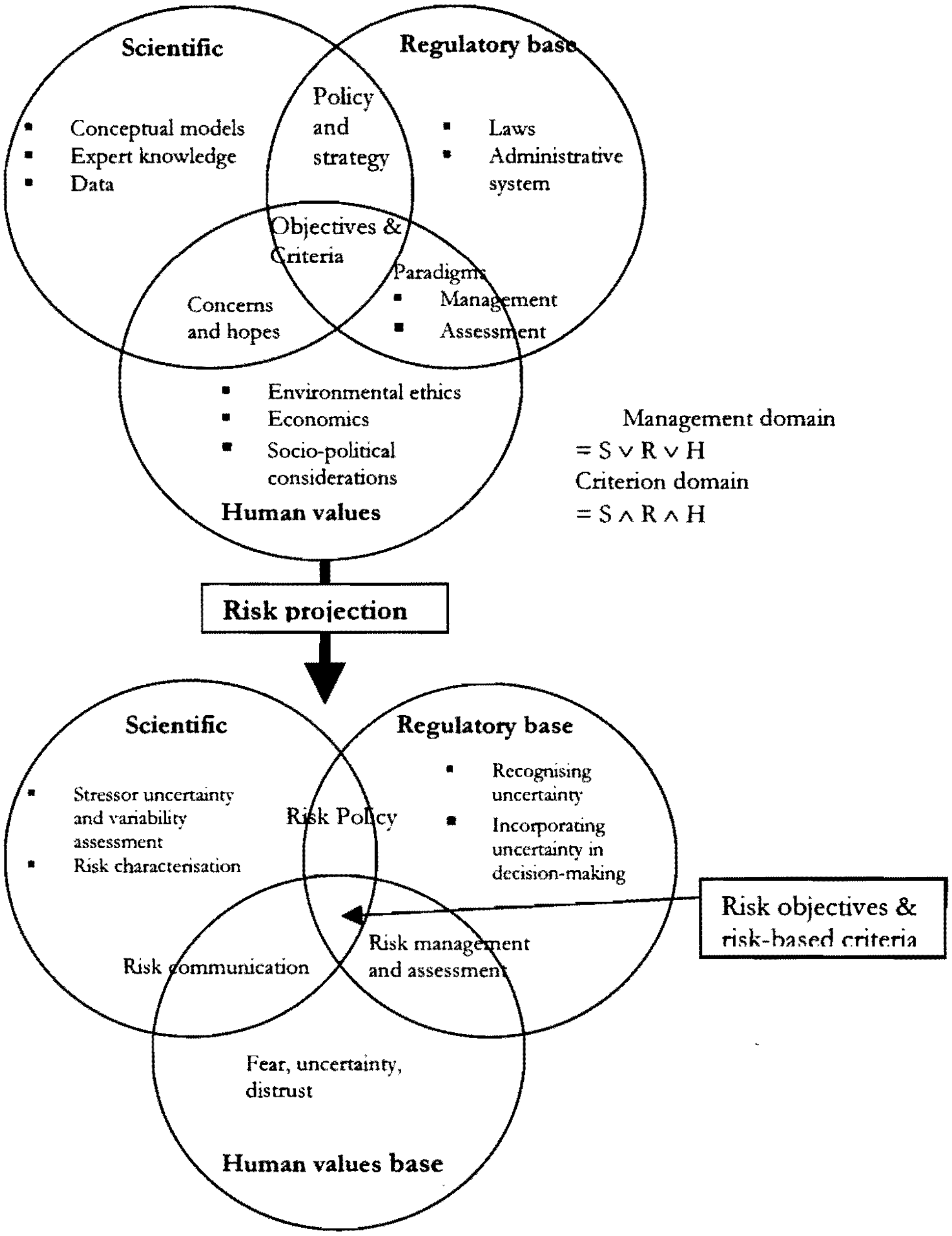

Figure 1.1 Some input domains of water resource management and how they relate to the application of risk-based decision-making

A conceptual model of the basis of management criteria is shown in Figure 1.1. The resource management domain is depicted as the conjunction of three of separate bases or domains, the boundaries of which are naturally fluid and fuzzy:

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b 15367757

1. The **scientific base** which deals with the gathering and systematising of ecological and other environmentally significant knowledge. This area will include most of the fundamental sciences like chemistry, physics, biology, geology and mathematics as well as some of the applied sciences like environmental chemistry, toxicology, hydrology, hydraulics, statistics, information technology, soil chemistry and physics, geomorphology, limnology and the like. These would be the group sometimes referred as the "hard" sciences.

2. The **regulatory base**, which deals with the laws and administrative systems, put in place both ranging from laws promulgated at central government level, down to operational rules of companies. These supply the infrastructure within which the day-to-day running of society takes place. It is likely that disciplines of macroeconomics, state administration and international affairs and political science would have an impact at this level.

3. The **human values base,** which deals with the way individuals and communities organise their lives and the way in which they view and would wish to manipulate their environment. Disciplines such as ethics (particularly environmental ethics), microeconomics and probably socio-political considerations would have an impact at this level. These are sometimes referred to as the "soft sciences".

Objectives and Criteria for resource management are impacted by all three domains and particularly by the interfaces between domains.

Policy and strategy is used here in the sense of technical policy and management strategy. These determine how some areas of uncertainty are to be handled in terms of, for example, assumptions that need to be made (e.g. when insufficient data are available, then a precautionary approach might be used or, to curb eutrophication, the use of phosphate builder in soaps might be phased out). The use of resource directed measures and source directed controls in water resource management are also a matter of management strategy.

The **management and assessment paradigms** stem largely from the way the human values interact with regulatory system, but it may (and should) be influenced by scientific knowledge. The assumption of a blanket precautionary approach, for example, may be influenced by a) a knowledge that the economy of the country as well as the socio-political situation will allow it, b) human environmental ethics dictate that "only the best is good enough for the environment" and in conjunction with this c) the legal system and regulatory framework require minimising possibly conflicting technical/scientific input. Furthermore, it might be required that an environmental assessment yield a clear acceptable/unacceptable answer because of the human mind's conditioning to see clear and unequivocal answers as the only expressions of certainty particularly in legal/ litigatory situations.

On the other hand, the interface between human domain and the scientific domain determines the **fears and hopes** both of the "lay" public and the "experts" who are, of course also human. This interfacial area also typically contains the area of science philosophy, which has an impact both on what is considered "good" science and what is considered "relevant" science.

> A criterion is a crucial component in regulatory administration that may have far reaching effects for the regulatee. While regulatory and scientific inputs may dominate in many cases, the derivation of viable criteria needs to recognise the importance of human values input. Practicable criterion derivation methodology should ensure that input from the human sciences can be accommodated in what might otherwise be a highly technical process.

1.4.2 BASIS FOR FORMULATING MANAGEMENT OBJECTIVES AND CRITERIA

Decisions and hence the formulation of the associated objectives and criteria in the management of the water resource could be:

1. **Bureaucracy driven**: i.e. management process is driven by the need for its own existence and is largely an administrative process. The bureaucracy driven approach is not a functional approach and when it does occur, it is more likely to be an artefact of a degraded administrative process and does not merit further discussion.

2. **Technology driven** i.e. the available technology and economics of the technology dominates decision-making while the effect of stressors on the system, is accommodated to the extent possible. The way in which effluent management criteria are set will therefore mirror the decision-making approach. Various technologies may be prescribed for emission impact reduction at source, such as Best Available Technology (BAT), Best Practical Means (BPM), Best Available Technology Not Entailing Excessive Cost (BATNEEC), as well as a number of other qualifying variants of the above (Foran and Fink, 1993). Presumably, the rationale in using technology-oriented decisionmaking (and effluent criteria) is that if the technology does not exist to effect a management action, that action is simply not viable. The disadvantage of such a

technology approach is that it does not necessarily achieve management goals and does not in itself supply the need for technology development.

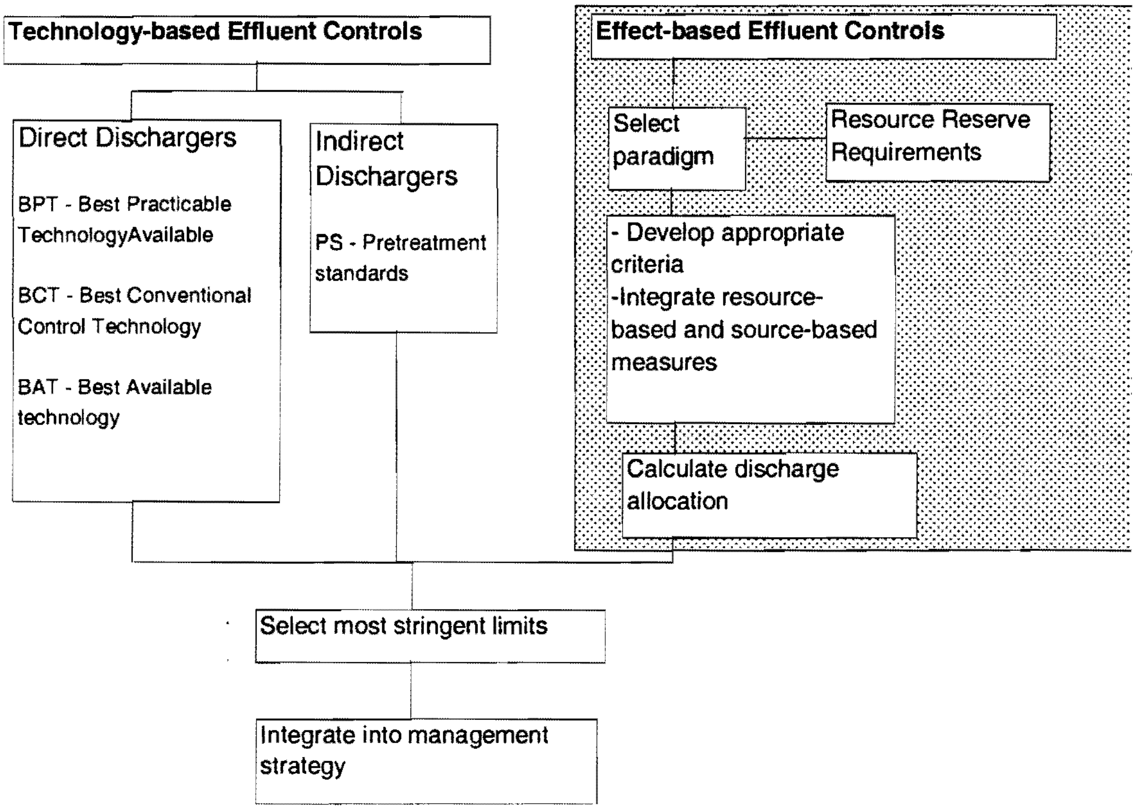


Figure 1.2. A diagrammatic of approaches to effluent management (adapted from Foran and Fink, 1993). The focus of this work concentrates on the shaded area. SDC's would be involved in the final step and could therefore draw on the output of this study.

3. **Resource driven** i.e. some valued function or process of the resource such as water use or economic activity rather than available technology drives management decisions. The effect of a stressor on the system dominates decision-making while technological limitations are recognised. Effect-driven decision-making (and effluent criteria) usually considers what the requirement is in-stream for some defined use of the water. This requires that some environmental quality objectives (EQO's) are set (Strortelder and Van der Guchte, 1995; Ragas, *et al.*, 1997). The EQO approach has been used in the UK while the technology based approach has predominated in countries such as Austria, Belgium, Germany and the Netherlands. In the USA, both approaches have been used in parallel (Foran and Fink, 1993). Technology based criteria are set and then the likelihood of violating EQO's are assessed. If the EQO's are likely to be violated

then the EQO approach is used to set criteria, otherwise the technology-based criteria are used. The latter two approaches are contrasted in Figure 1.2.

From the point of view of the resource management to achieve the Reserve goal, it would be preferable to follow an effect-based (e.g. environmental quality objectives or EQO) approach rather than a technology based approach. This has been suggested for use in South Africa (Van der Merwe and Grobler, 1990). The goal of the NWA is to achieve a specific effect, i.e. to maintain sustainability in the ecosystem. Consequently, the EQO approach has to be adapted to the characteristics of the ecosystem and ecological processes, as well as the needs of the catchment, particularly:

- It needs to recognise that not only the chemical and physico-chemical composition of water is involved, but that a diverse range of stressors might be involved,
- There is a natural variability in environmental conditions (including a specific frequency of extreme events such as floods and droughts), that is not only innocuous but necessary (CSIR, 1989).
- While resource objective driven decisionmaking may supply the impetus for technology development, it is still dependent on the technology necessary to achieve these goals. This implies that a purely effect-driven approach to setting EQO's may not be viable. The limitations and implications of underpinning technology need to be recognised.

1.5 THE DIVERSE-STRESSOR-MULTIPLE-SOURCE (DSMS) PROBLEM

While stressor-specific point-source criteria or standards are administratively advantageous, it can be shown (Part 2: Paper 1) that it is no guarantee of desired in-stream effect. For this reason, the concept of in-stream water quality objectives was used. The in-stream objective could be set to correspond to the level of a water quality variable which is expected to provide the desired level of protection (with perhaps a safety factor added). Establishing the end-of-pipe criteria corresponding to these objectives necessitates the use of waste load allocations (WLA's). The total load corresponding to the objective concentration (in the case of stressors in solution) can then be apportioned among the sources of such stressors. However, in terms of the Reserve required under the NWA, the conventional WLA to stressor specific water quality objectives is at a disadvantage because of:

- The **additivity effect** of a number of **similar stressors**. E.g. the combined effect of a number of different toxic substances which are discharged to a river (each of which complies to its own particular acceptable effect concentration) may be greater than acceptable due to some form of additive or supra-additive (or even synergistic) effect. This problem on its own is not insurmountable since stream objectives may be adjusted to accommodate this phenomenon but it becomes administratively cumbersome.

- The **diverse-stressor (DS) problem**. Even when additive effects among toxicologically similar stressors are accounted for, estimating the combined **effect of dissimilar stressors** may be impossible. The action of the stressors may be mechanistically dissimilar although the final effect may be the same. A WLA in itself cannot overcome this problem.
- The **diverse-stressor-multiple-source (DSMS) problem**. When a number of heterogeneous stressor sources have to be accommodated, this exacerbates the DS problem. Now a common basis for expressing impacts is called for in order to optimise the apportionment of stressor attenuation. Stressor metrics (such as concentration and flow) is no intrinsic **common basis for comparison** on which WLA may be based. When apportioning toxic substance load, nutrient load and flow deficiency (all of which may result in ecosystem stress), for example, the stressors are dissimilar both in units of measurement and mechanistically. Not only is the effect of diverse stressors not accounted for, but the allocation of the stressor load among different sources can lead to an infinite number of combinations of stressors that are all equally valid.

Fundamentally, the problem described here is that the WLA tends to be dominated by the stressor rather than by its effect. Changing from an stressor- to an effect-oriented approach may solve the problem since a fundamental rationale of water resource management (or any other resource management for that matter) is to achieve a specific goal by managing the inputs.

1.6 RATIONALE FOR THE USE OF RISK METHODOLOGY

The rationale for using risk-oriented methodology is argued in Part 3, Paper 1. Some of the main points are listed here.

1.6.1 A RISK APPROACH

A risk approach is used here as a counterpoint to a hazard approach to resource management. A hazard in this context refers to the potential that a stressor has to cause some unacceptable effect. The SAWQG criteria are examples of hazard-based criteria.

HAZARDS AND HAZARD-BASED CRITERIA

The criterion derivation process for the SAWQG's used toxicity data, but by assumption specific benchmarks of effect (such as LC50 values in the case of the Acute Effect Value or AEV) were selected as the basis for criterion derivation. The AEV would be an indication of maximally acceptable hazard. All the uncertainty relating to the data and derivation process has been discounted by precautionary assumptions (Roux, *et al.*, 1996).

By definition any single hazard-based criterion recognises only one type and level of effect (e.g. mortality at the 50^{th} percentile in the case of the AEV). Consequently only the stressor and its characteristics are considered variable. A hazard-based criterion would therefore typically be a stressor value corresponding to a level of acceptable effect (e.g. the general AEV for cadmium in moderately hard water is 6µg/l). There is no indication of how the hazard changes as the stressor value changes, for example. The hazard either exists or it doesn't. So, when apportioning the load, using a hazard criterion gives no indication how disastrous it would be if the objective were temporarily exceeded by 10%, or 20% or even 50% This would normally call for expert opinion and it is a soluble problem, but the solution is not implicit in the problem formulation

If the assumptions in the derivation process are explicitly precautionary, then the criteria are useful in setting the most stringent on a stressor-by-stressor basis. As such, they may define the most conservative end of the management objective spectrum.

> Hazard-based criteria are useful management tools inasmuch as they may represent the precautionary objectives for resource management. However, they may lack the flexibility necessary for the management of diverse stressors in a multiple source environment.

The type of criterion is also closely associated with the paradigm in which it is used (See the quantal assessment paradigm (QAP) and the continuous assessment paradigm (CAP) described in Appendix A1.2). Hazard-based criteria are necessarily associated with the QAP (although the use of the QAP does not necessarily imply the use of hazard criteria). While it is useful to have fixed values of variables to assess situations for law-enforcement, it must be recognised that this does not make the best use of all the available scientific information.

RATIONALE FOR THE RISK APPROACH

In characterising the Reserve and managing for its sustainable use, some fundamental characteristics of the ecosystems and ecological assessments need to be noted:

1. There is an innate and practically irreducible inter- and intraspecific variability in biotic response to a given stressor as well as in many other aspects of in biotic systems (O'Niell *et al*, 1979;

Kooijman, 1987, Levine, 1989; Brown, 1993). (These concepts are discussed more extensively in Chapter 2.)

2. In many natural ecosystems there is a dearth of detailed data about structure, function and composition that adds to the overall uncertainty regarding ecosystem models and their predictions, which limits the scientific certainty about any biotic system and its responses.
3. The response of organisms to stressors is normally continuous and discontinuities are normally an artefact of the scale or means of observation (notwithstanding the possibility of a threshold of effect). Generally, there are no natural discretisations in the continuum of response..

The consequence of this is that a deterministic, quantal view of management actions and their consequences may be inappropriate. A more probabilistic, continuous approach as typified in the continuous assessment paradigm (CAP, see Appendix A1.2.1) is indicated. Risk is a suitable basis for ecological assessment in the context of the Reserve and RDM's since it:

- Is by definition, is a probabilistic expression and therefore caters uncertainty and variability explicitly (See e.g. Bain and Engelhard, 1987).
- Allows for a CAP (Suter, 1993) since it allows the use of all the stressor response data as well as the exposure data.
- Is explicitly effect-based as it requires an explicit end-point, which could incorporate the human concerns.
- Probability theory allows for events (such as the occurrence of a selected end-point dependent on the occurrence of different stressors) to be partitioned into component events (such as the occurrence of the end point dependent on single stressors or selected groups of stressors). A theoretical underpinning exists for establishing the relationship between the main event and the component events (see Chapter2).

It is postulated that risk as a more suitable basis on which to base objectives and criteria related to resource management compared to hazard, since the characteristics of risk is better suited to the ecological assessment domain than hazard. This supposes that risk objectives analogous to hazard objectives can or have been set.

1.7 GOAL AND OBJECTIVES

The aim of this study is to introduce, at a conceptual level, the use of risk or risk-related methodology to solve the DSMS problem (in 1.5 above) in the context of the ecological Reserve required under the South African NWA or in any situation where risk objectives can or has been set for a water resource.

In particular, source-specific criteria are envisaged that correspond to ecological risk objectives set for the water resource, while at the same time recognising that technological or other factors may determine the level of acceptable stressor reduction.

These source management criteria are not meant to supplant any other resource criteria (such as the SAWQG criteria for the protection of the aquatic ecosystem). Such water quality objectives may still form the basis source-specific waste load allocation of individual stressors where appropriate. The risk-based source-specific criteria will likely only be applied in a catchment management context and only when: a) there are indications that several diverse stressors may all contribute to an impact on the water resource, or b) there is conflict among source managers and regulatory authorities.

1.7.1 GOAL

The problem to be solved can therefore be formulated as: **Find a rational means to derive stressor-source management criteria that give effect to the Reserve concept in a catchment when there are multiple (diverse) stressors originating from a number of identifiable and manageable sources present in a catchment, taking into account that management criteria have definite socio-economic as well as technical implications.**

1.7.2 OBJECTIVES

In order to achieve this goal, the following objectives need to be met:

- The formulation of end-point projection problem. How to relate the likelihood of effect at a higher ecological level when only data for the estimation of a lower end-point is available (Chapter 2).
- Formulating stressor-response relationships. The estimation of the likelihood of effect is a fundamental requirement of the ecological risk (Chapter 2).
- Solving the diverse stressor problem. How to estimate likelihood of a specific effect when diverse stressors occur together. This amounts to a mathematical formulation of the ecological risk characterisation step in the ERA process (Chapter 3).
- Formulating DSMS problem as an optimisation problem and solving the optimisation problem (Chapter 4).

CHAPTER 2

BACKGROUND AND THEORETICAL CONSIDERATIONS

If a man will begin with certainties he shall end in doubts; but if he will be content to begin with doubts he shall end in certainties – SIR FRANCIS BACON

2.1 SUMMARY

In this chapter the difference between ecological risk assessment (ERA) and ecological risk based management (ERBM) is investigated further. The effect assessment phase would include formulating a stressor-response relationship (SRR).

Two major issues in formulating the SRR are: a) deriving a relationship between the likelihood of observing an end-point at higher (both conceptual and organisational) levels when only lower level data are available, and b) informing the SRR's.

The end-point projection problem is formulated in both probabilistic and possibilistic frameworks. The obvious point is demonstrated that the confidence in the risk with higher-level end-point cannot be greater than the risk predicted from lower level data.

Data for informing toxic SRR's will need to be derived from toxicity bioassessment, but careful attention needs to be given to factors such as level of organisation of the end-point and time variable toxicity levels.

Flow and habitat SRR's are likely to depend on expert opinion. It is therefore necessary to establish methodology by which to update the SRR's from field observations. Dempster-Schafer and other updating methods may be applicable.

2.2 INTRODUCTION

2.2.1 ECOLOGICAL RISK ASSESSMENT VS. ECOLOGICAL RISK-BASED MANAGEMENT

ECOLOGICAL RISK ASSESSMENT

Risk assessment is a well-established tool in both economics and engineering. The application of risk assessment to ecological assessment, ecological risk assessment (ERA), is a tool in environmental management. It is mostly used in the context of predictive risk assessment when a stressor is given. The framework and techniques of ERA have been widely used and are well known (Suter, 1993; Crouch, *et al.*, 1995; EPA, 1996; EPA, 1998). A simplified process diagram for ERA appears in Figure 2.1 while Figure 2.2 adds some more detail to show the interrelationship between ERA and risk management.

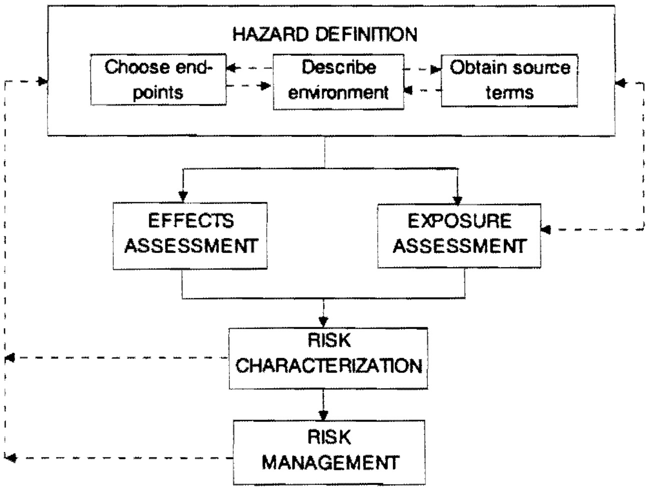


Figure 2.1. A simplified diagrammatic representation of the process of ecological risk assessment illustrating the main steps. The dashed arrows indicate feedback loops in the risk assessment paradigm. (From Suter, 1993).

ERA provides a structured methodology to formulate the societal values in measurable end-points and then to assess the likelihood of the occurrence of this end-point (EPA, 1998). The expression of risk in terms of likelihood stems explicitly from recognising the impact of uncertainty and variability (see 2.3 below) on the outcome of the assessment. This stands in contrast to some forms of environmental impact assessment that takes great pains to enumerate the potential impacts, but stops short of making an explicit assessment of the impact of uncertainty and variability on the overall situation assessment (DEAT, 1992; DEAT, 1998).

ERA has been used extensively in the management of stressors (pollutants) in the environment. It supplies a relatively objective means to compare different stressors, sources or treatment techniques. The methodology incorporates the best available knowledge on the source, environmental partitioning, and ecotoxicology of a stressor, the ecology of the receiving environment as well as societal concerns and issues and expresses it as a risk.

The expression of risk as used commonly in ERA involves some concept of likelihood of an effect on a target entity in the ecosystem, while the dimension of the stressor does not necessarily have to appear. For example the result of an ERA might be: "The probability of the loss of 10% of species due to stressor A is 0.01 while the probability for the same end-point due to stressor B is 0.2". In this way, it supplies a common basis for the comparison of otherwise dimensionally incompatible stressors. At the same time it is also a basis for communication of a rather technical process with a (possibly) technically illiterate or semi-literate audience.

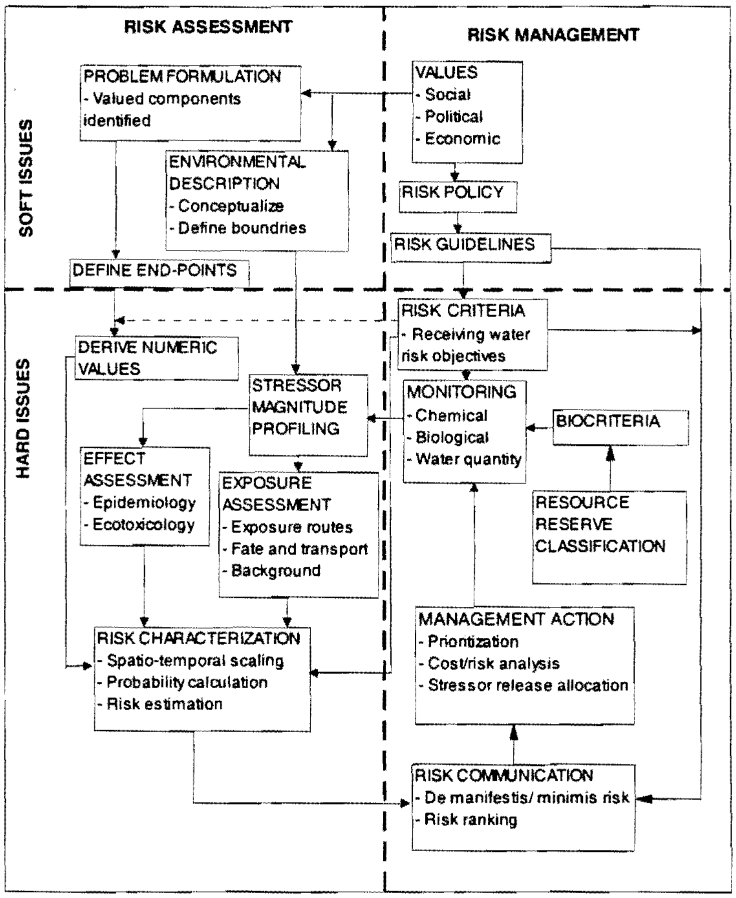


Figure 2.2 A more detailed analysis of the hard and soft issues involved in predictive ERA and its relation to risk management.

The rationale for applying ERA stems from the implicit question: "If stressor X occurs and effect E is the allowable effect, what is the likelihood (perhaps expressed as probability) that X will result in E?" In this case the stressor will be characterised by measured or predicted values of X.

2.2.2 NOTES ON CONVENTIONAL ERA

The main features the ERA process (Figure 2.1) include:

1. The **hazard definition** or (**problem formulation**) phase where an **end-point** for the assessment is selected, the **environment** in which the assessment is performed is described and, in general, the **stressor source** is characterised. The end-point includes both a **target ecological entity** and a **specific effect**.
2. The **effect assessment** phase in which (among other things) the relationship between the magnitude of the stressor and the likelihood of observing the end-point is identified.
3. The **exposure assessment** phase, where the likelihood of exposure of the target entity to the stressor is characterised.
4. In the **risk characterisation** phase the effect and exposure data is convoluted to obtain a quantitative or qualitative risk estimate (among other things).
5. The risk estimate is fed back to the **risk management** phase where the risk assessment request most likely had its origin.

With regard to the **hazard definition** or problem **formulation phase** it is noted that:

(a) An assessment end-point is required which, whatever that target entity is, has unquestionable or at least consensus value within the decision-making group (the upper right quadrant in Figure 2.2).

(b) Explicit provision is made for ecological models in the problem formulation phase of ERA that ensures that all routes of exposure to all relevant ecological compartments are addressed (Suter, 1996).

(c) Conceptual model development, which consists of formulating and contextualising the risk hypotheses. Risk hypotheses (*inter alia*) are assumptions about the consequences of risk assessment end-points and may be based on theoretical models, logic, empirical data or probability models. In complex systems, they are likely to be strongly dependent on expert judgement. The point of these hypotheses is ultimately to structure the analysis. It provides a link between the actual knowledge and problem it sets out to solve. In addition, they are useful in accounting for and characterising the uncertainty in an assessment.

With regard to the **effect assessment phase** it should be noted that:

i. All the available data should be used to establish the relationship between the selected end-point and the stressor occurrence

ii. All lines of evidence should be investigated. This might include information from laboratory studies, direct field observation of stressor-target entity interactions at the risk assessment site or inferred interaction from other suitable sites.

iii. All of the above can in principle be synthesised into a stressor response relationship (SRR), which is an expression of the functional relationship between the level of a stressor and the expected impact on the end-point effect on the target ecological entity. This might, for example, be expressed as a mathematical function or a rule base.

With regard to the **risk characterisation** phase (Suter, 1995):

i. The simplest form of expressing risk is by a point estimate such as the ratio between the expected stressor level (ESL) and some benchmark effect level (BEL). In this form it takes no cognisance of the uncertainties in variability involved in the assessment.

ii. Taking uncertainty into consideration, risk could be expressed as Likelihood (ESL > BEL).

iii. There does not appear to be a formal, generally accepted formulation of the relationship between risk, the SSR and the stressor exposure distribution.

2.2.3 RISK-BASED MANAGEMENT UNDER THE NWA

If risk is to be used to harmonise RDM class goals with SDC criteria, then it is implicit that a risk should be given as a goal. The RDM classification protocol contains the sense of risk implicitly. The basis for classification is the risk of destroying the Reserve. This risk is here defined as the **resource class risk objective**.

In the process of establishing the relationship between RDM's and SDC's it is necessary to establish the characteristics of the stressor given a risk objective. This process will be referred to as ecological risk based management (ERBM). Here the implicit question is somewhat different: "If effect E with likelihood R is all that can be allowed, what should the characteristics of stressor X (perhaps expressed as probability) be to accomplish this?" The ERBM process is very similar to the ERA process (Figure 2.1) except that the risk characterisation step and the flow of information is essentially the reverse of that for ERA (Figure 2.3).

When several stressors occur together in a water resource for example, available methodologies allow for a risk assessment for each individual stressor to be performed. It appears to be feasible to make use of the likelihood expression of risk to obtain an indication of the likelihood of the end-point phenomenon. With a management goal oriented choice of end-point, the integrated risk with respect to this end-point may then be a rational basis for apportioning the use of the water resource.

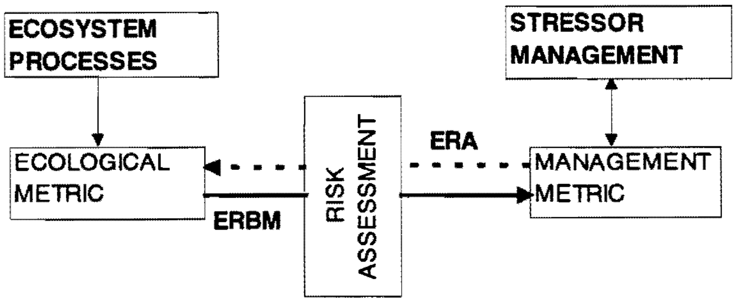


Figure 2.3 A comparison between the ecological risk assessment flow of information (dashed lines) and that of ecological risk based management. Some form of risk assessment framework remains the interface between the management metric (such as stressor release rate) and the ecological metric (such as sustainability or resilience). The risk assessment interface for ERBM *is expanded in Figure 2.4*

In its most fundamental form, a risk numeric value is calculated from some form of convolution of an effect likelihood expression (e.g. a probability distribution) and a stressor occurrence likelihood expression. If risk is expressed probabilistically, then deconvolution for the ERBM process could be very difficult. It would involve calculating every combination of effect probability-stressor probability that could result in a particular risk probability.

It can be concluded that:

(From Chapter 1) in the application of risk methodology under the NWA both the target ecological entity and the end-point is fixed. The target ecological entity is the ecosystem and the end-point is sustainability

The approach in ecological risk-based management (ERBM) is in a sense the converse of ERA. The point of risk-based management is to assess the level of stressor corresponding to an accepted level of risk.

In both ERA and ERBM stressor response relationships (SRR's) are important. A formalised structure for relating the regulatory end-point to the experimental/ observation level end-point. The ways in which the SRR is informed from observational data needs to be considered.

For ERBM under the NWA it is necessary to be able to express the aggregate risk. A mathematical expression of aggregating individual stressor risk is needed.

An expansion of a generic ERBM process might be summarised as shown in Figure 2.4. This study concerns itself with the shaded areas in this diagram.

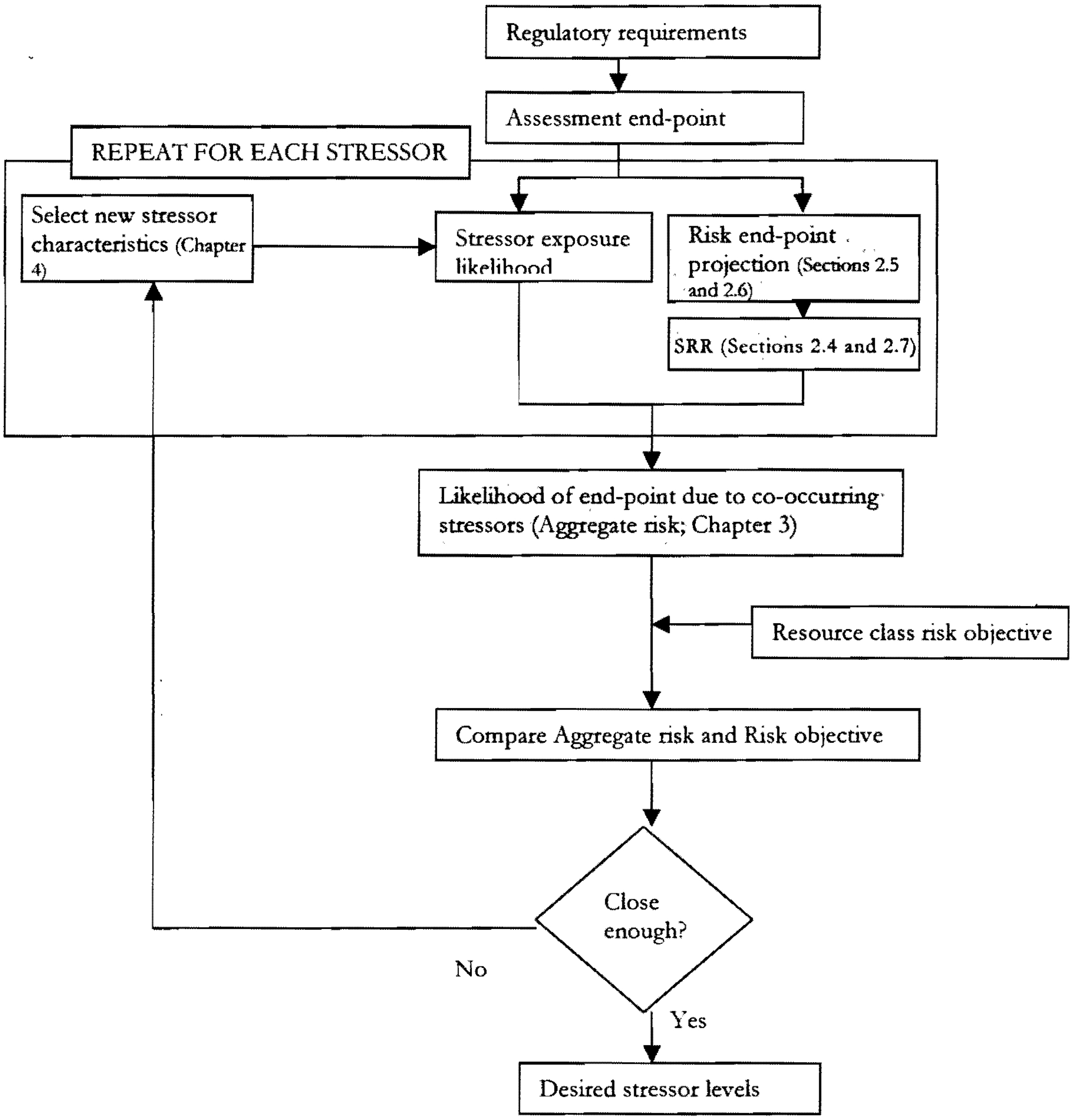


Figure 2.4. Diagram of a generic application of ERBM framework showing how aspects of the ERA process are used. This study concerns itself with the shaded areas in the diagram.

2.2.4 RISK AS LIKELIHOOD

Although many of the formal definitions of risk (such as those referenced under definitions) emphasises the probability aspect of risk assessment, the general problem is in estimating likelihood of adverse effects (Suter, 1995). The term "probability" has come to be associated in technical literature with precise but stochastically distributed observations. In the management of ecosystems this definition cannot always be met (See Chapter 1). System specific knowledge may at times be imprecise or uncertain and not necessarily influenced by randomness. In view of the discussion in 2.3 below, it is fitting that the term "likelihood" rather than "probability" is used in referring to ERA in general.

2.3 UNCERTAINTY AND VARIABILITY

It has been recognised that the rationale for risk assessment stems from the need to incorporate the effect of uncertainty and variability on decision-making (Frey, 1993; CRARM, 1997; EPA, 1998).

Colloquially, variability may be seen as a source of uncertainty in an estimation. Within the risk assessment community there is a distinction drawn between uncertainty and variability (Frey, 1993).

The phenomena referred to in the conceptual Reserve-related end-point formulation may be subject to either or both uncertainty and variability. With reference to ecological risk assessment, it has been recommended that uncertainty and variability be separated to provide greater accountability and transparency in a probabilistic assessment (USEPA, 1997b).

2.3.1 VARIABILITY

Variability is recognised as a natural characteristic of biota (e.g. Brown, 1993, Grimm and Uchmanski, 1994, Kooijman, 1994). Several forms of variability could be encountered. There is variability in the individual response of the biota to a given stressor exposure (Hathway, 1984) which is evident in the classic dose response curve of toxicology. Other stressor-response curves may, in principle, appear similar although the curves need not necessarily be strictly monotonic. Although these functions may not necessarily be measurable in controlled laboratory experiments, a combination of field observation and expert interpretation is likely to provide an estimate of the stressor response relationships. In this regard the use of a Bayesian statistical approach rather than a strict frequentist approach may be indicated (Frey, 1993).

Variability has the following characteristics:

- It is **inherent characteristic** of the system being observed.
- It stems from an underlying **stochastic mechanism** in which the outcome of the process is essentially precise in nature but randomly distributed over an outcome space.
- The **laws of probability** apply to variable quantities. Whether explicitly or implicitly, the concept of the **repeated experiment**, which is at the heart of statistical theory, can be applied to variability.

EPISTEMIC INTERPRETATION OF VARIABILITY

In ecology there is seldom a situation where experiments can literally be repeated. As pointed out by Thomas (1995), for one thing, time will have elapsed. In dynamic systems, such as ecosystems, this will mean that the system has already moved to another point in its state space, and that in principle, no experiment can be exactly duplicated. However, there may exist an **experimental morphology**, which, for the observer's purposes, is repeatable.

> Example: Thomas (*op. cit.*) quotes the mathematician Cramer in describing the assessment of the probability in 1944 that the Second World War would come to an end. Although this war was unique in history, there were elements with regard to the strategic positions of the various armies, the morale of the troops, the resources available to the warring factions etc., that could be compared to those in other conflagrations, and which would lead the observer to estimate the likelihood of an end to hostility.

Table 2.1. Some of the characteristics of uncertainty and variability with particular reference to ecological models (based on Frey, 1993 and USEPA, 1997b).

Characteristic	**Uncertainty**	**Variability**
Source	Lack of empirical knowledge of the observer or imperfect means of observation.	True heterogeneity inherent in a well characterised population
Impacted by:	Model uncertainty • Model structure • Range of conceptual models Parameter uncertainty • Random error due to imperfect measurement • Systematic error (bias) • Inherent stochasticity or chaos • Lack of empirical basis • Unverified correlation among uncertain quantities • Expert disagreement on data interpretation	Individualism in response Lack of representative data Aggregation dimension (e.g. time or space)
Encoding	(Bayesian) Probability distribution	Frequency distribution
Effect of more data	Reduces	Unchanged but more precisely known
Applicability of standard statistical data analyses	Understated (due to focus on random error to the exclusion of bias introduced by variability)	Overstated (due to inclusion of measurement error)

2.3.2 UNCERTAINTY, VAGUENESS AND AMBIGUITY

UNCERTAINTY

It is necessary to distinguish between uncertainty and variability since it has an impact on the way in which likelihood is expressed and interpreted. The likelihood of a phenomenon of the model may be influenced by two broad categories of causes: epistemic uncertainty or systemic uncertainty.

- **Epistemic uncertainty** refers to the situation where the knowledge about, and hence the description of the system is uncertain

- **Systemic uncertainty** refers to the situation where the system itself is uncertain in its definition even though the tools for its description are precise. A comparison between uncertainty and variability is made in Table 2.1.

Essentially, what distinguishes uncertainty from variability is the lack of a stochastic basis. Uncertainty is a characteristic of an observer rather than of a system and stems from a lack of knowledge. Frey (1993) resolves two kinds of uncertainty: model uncertainty and parameter uncertainty.

- The model uncertainty in the case of ecosystem models is due to imperfect knowledge of a specific ecosystem's processes and mechanisms. There may be several options that may be conceptually valid based on the study of other similar ecosystems or mechanistic models.
- The stress responses may be quite precise, but the discrimination among the model choices may be blurred. This phenomenon is exacerbated by parameter uncertainty. Even when the specific model used to predict effects is known, very often the parameter values are wholly or partially unknown or the number of parameters is unknown. The sources of parameter uncertainty are listed in Table 2.1. It is apparent the variability as used above may be a subset uncertainty.

In many cases, it is possible to extrapolate from simple systems, such as laboratory test systems, to ecosystems on various bases, but with a significant loss in confidence (See Table 2.2). However, much of the work done on extrapolation and projection is only applicable to the effect of toxics. Characteristic of these extrapolations is the dependence on system specific knowledge and the rapid increase in uncertainty.

VAGUENESS AND AMBIGUITY

In the description of variability and uncertainty in Section 2.3 above, the outcome of stress is precise although not deterministically predictable. In principle at least, an experiment can be conducted which will elucidate the effect of a stressor on an individual organism (for example) and that will uniquely define that particular individual's response. Repeating the experiment on a large number of individuals will characterise the expectation of response better but it will not remove the variability of the population response.

In contrast to variability, the observer's personal sense of confidence in assessing the outcome of stress applied to an ecosystem may also be hampered by uncertainty, vagueness and ambiguity. These differ from variability in that, while variability is a characteristic of the system, uncertainty, vagueness and ambiguity is a characteristic of the observer.

In contrast to uncertainty, vagueness relates to the precision with which inputs and outputs in the predictive or analytical process is known. In the context of the NWA, terms such as "sustainable" are left undefined. The definitions in 2.4.2 derived from literature sources, are vague. In addition, qualifiers such as "adequate sustainability", "adequate resilience" and "massive abnormal mortality" are functionally vague terms but are nevertheless descriptive. The choice of phraseology is intentionally vague as the values by which it is characterised is highly site- and situation-specific. A term such as "adequate" as a qualifier for sustainability may take on a range of values as opposed to the qualifiers "low" or "high". But the interpretation of the term is qualitatively clear and its implications scientifically interpretable.

2.4 STRESSOR RESPONSE RELATIONSHIPS FOR ERBM

As noted in 2.2 above, a SRR is a functional relationship between an end-point and the magnitude of the stressor. In view of the impact of uncertainty and variability as discussed in 2.3, it may in general be impossible to specify ecological effects deterministically. Consequently, an ecological SRR may at best be expressed as a likelihood that a selected endpoint may be observed. For ERBM decisions to be scientifically tenable and legally valid, the SRR should:

a. Refer to the regulatory end-point rather than a laboratory or other field observational end-point (i.e. the Response Inference problem referred to in 2.4.1), and
b. Make the best possible use of all relevant information. This involves formulating the Response Inference on a basis suitable to the data at hand (2.5 and 2.6).

2.4.1 THE RESPONSE INFERENCE PROBLEM

The general form of this problem can be described as follows: "You (the assessor) are required to make a pronouncement about the impact of a stressor at a higher level of organisation (such as at the ecosystem level) and at a conceptual level (in terms of sustainability for example) which is far removed from the experimental data You have available". The problem, therefore, concerns both organisational and conceptual scaling of response end-points.

THE ISSUE OF SCALE

Figure 2.5 illustrates the problem with scale in the estimation of ecological stressor-response relationships. The difference in scale results in an incongruence between the level of the data available for making decisions and the level of the impact of those decisions.

Data scale

In many cases estimates of effect are based on laboratory data generated from experiments performed to observe the change in physiological functions of individual organisms (e.g. measured as change in reproductivity, cessation of vital function, change in behaviour, etc.) on exposure to a stressor. It estimates effects at a scale of perhaps a few millimetres to perhaps tens of metres (in the case of micro- or mesocosm experiments) and hours to perhaps a few months (Sugiura, 1992; Graney, *et al.*, 1994). The regular experiments may therefore cover the domain of spills or short-term pollution incidents.

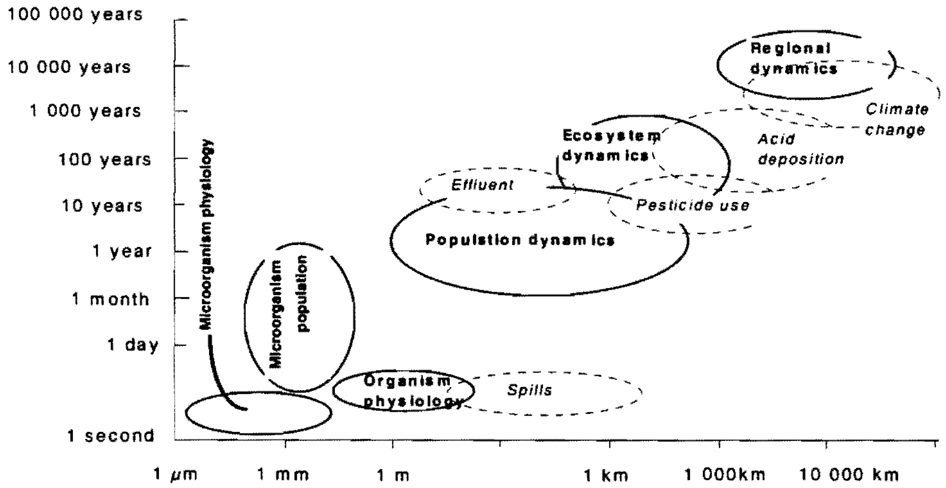


Figure 2.5 Temporal and spatial domains of some ecological factors and typical stressors (adapted from Suter, 1993).

Management scale

The greater problem for South African surface water management, where the major source of flow in the dry season is comprised of effluent, is that its impacts occur in the spatial domain of tens of meters to several kilometres and the temporal domain of several decades.

Regulatory scale

The National Water Act sets a goal (ecological sustainability) at ecosystem scale, for which the responses are in the spatial domain of a few meters to hundred of kilometres in the temporal scale of a few years to centuries. In many cases it is not possible to conduct experiments at the required spatial and temporal scale to estimate stressor response directly. There is a need to perform extrapolations from the observational scale to the required scale (Suter, 1990; Landis and Yu, 1995). When extrapolations such as those in Table 2.5 (Section 2.7) are used, it should be carefully noted whether the extrapolations refers to both spatial and temporal scaling.

Scaling impacts or responses over different levels of ecological organisation, spatial and temporal domains necessarily means that there is a loss in confidence. To address this uncertainty systematically, a model of relationships of various end-points pertinent to the aquatic ecosystem is needed.

2.4.2 ECOLOGICAL PHENOMENA

A distinction is now made between issues (such as "sustainability", "integrity", etc.) and end-points, which specifies some characteristic of the issue (such as "loss of sustainability"). It is proposed that when higher level issues, such as sustainability are addressed, there are natural "milestone issues" that can be defined in terms of biological descriptors such as "integrity", "biodiversity", etc. These issues can be associated end-point events or phenomena, which would be described as the attainment (or conversely, the loss) of such a "milestone event".

In an assessment of risk at this level, the term "likelihood" essentially expresses confidence that such an event can (or has) taken place. Each phenomenon or event may, in principle, be arrived at in many mechanistically different ways, each of which influences the likelihood that the phenomenon could be observed. However, the likelihood of observing a phenomenon is not dependent on knowledge of the mechanistic detail, but rather on the epistemology of the event.

A phenomenological rather than a mechanistic basis is chosen to facilitate the incorporation of expert judgement and observational data at higher levels of ecological organisation (where mechanistic knowledge is often lacking). It is assumed that a phenomenological model should have the following characteristics:

A. The phenomena should be linked by logical inference.
B. Methodology should be available to assess the state of the phenomena, which implies that there should be metrics for the state (e.g. see Table 2.2). The risk is then the expression of

the likelihood that a given set of state-descriptors characterising the phenomenon is attained or lost.

C. The phenomena should be chosen at an organisational level suitable to the assessment (Figure 2.5). As the state of mechanistic knowledge increases, the phenomena could be resolved further until, conceptually, phenomena at molecular level or lower can be related to the higher level phenomenon. If no measurement end-point exists at the level of the assessment, the assessment should not be changed to suit the end-point. Rather the model should be used to emphasise the information need. Failure to do this results in a false sense of confidence.

Table 2.2 Indicator variables for assessing biodiversity at three levels of organisation. (Based on Noss, 1990 and augmented from Pratt and Cairns, 1996, Karr, 1993)

Level	Indicators		
	Composition	**Structure**	**Function**
Community/ecosystem	Identity, relative abundance, frequency, richness, evenness, diversity of species or guilds, succession	Abundance, density and distribution of key physical features and structural elements of rivers, Food web assembly	Biomass productivity, parasitism, predation rates, colonisation end local extinction rates, patch dynamics, nutrient cycling rates, biogeochemical cycles
Population/species	Absolute or relative abundance, biomass, density, primary production and primary and secondary consumption	Dispersion (micro distribution), range, population structure (e.g. age ratio), habitat variables (as above)	Demographic processes (e.g. fertility, survivorship), population fluctuations, physiology, life history, individual growth rates
Genetic /cellular	Allelic diversity	Census and effective population size, generation overlap, heritability	Inbreeding depression, gene flow, mutation rate, selection intensity, and photosynthesis.

MODEL POSTULATES

The conceptual model is based on the following postulates:

1. The **reference state** for the model is the **pristine system**. It is implicitly assumed that the reference state's only fixed characteristic is that it is pristine, but that the values of the descriptors could be spatially and temporally variable. There exists a pristine **pattern of natural extreme events** such as droughts or floods which are not stressful and which may be necessary (due to adaptation) in arid or semi-arid regions such as South Africa (DWAF, 1987; Davies, *et al.*, 1994).

2. The quest for the maintenance of sustainability only arises because there is real or implied **anthropogenic threat** to the system. Sustainability is not defined for a system not subject to any threat of anthropogenic stress.
3. The phenomenon "sustainability is maintained" occurs only if the phenomenon " suitable level of integrity is maintained" occurs. The state of **integrity** of the system is determined by its **state of biotic integrity, habitat integrity and the natural temporal patterns of extreme events**. For integrity to be maintained neither habitat diversity, nor biodiversity nor the natural temporal event pattern should have been disrupted (Odum, 1985; Pratt and Rosenberger, 1993; Naeem, *et al.*, 1994).
4. **Biodiversity**, in terms of the **composition, structure** and **function** of the system (each at several levels of organisation from molecular to landscape level) is defined in relation to the state of these components in a pristine system. Biodiversity as a variable indicating stress is subject to an interpretation of the **individual importance** of species. Redundancy is possible or even probable in an ecosystem and the real question is how much diversity could be lost without pushing the system to the edge of some irreversible, catastrophic change (DeLeo and Levin, 1997). The conservative assumption would be that **all species are equally important** (rivet popper hypothesis) (Walker, 1991).
5. For biodiversity to be maintained, neither the structure nor the function of biota should have been impaired. Any such **impairment**, by definition, implies **loss of integrity**.
6. Rapport, *et al.* (1985) point out that integrity is lost more easily in a system subject to **constant low-level stress** compared to a system subject to **infrequent high intensity stress**. Qualitatively this is modelled analogous to the model of reversible toxic effect (e.g. Hathway, 1984; Verhaar *et al.*, 1999; Freidig *et al.*, 1999). The absence of stress is interpreted to mean that, either or both the level of the stressor was not high enough, OR that the duration of exposure to the stressor was too short to make any impact.
7. An ecosystem is assumed to be impacted by **chemical water quality** or **physical quality** of its **habitat**, or by the stress related to the **flow rate** of the water comprising its physical habitat or by the presence of **exotic biota**.
8. The long-term effect of stressors is also dependent on the **availability of refugiae** from which the population numbers can be replenished. If no such refugiae exist, then the **population viability** is dependent on **sufficient numbers** to maintain its status despite natural mortality and normal biotic interactions such as predation and competition. The **precautionary approach would be to assume that no refugiae exist**, but this restriction could be lifted on a site-specific basis.

2.4.3 A CONCEPTUAL MODEL FOR END-POINT SCALING

PHENOMENOLOGICAL INFERENCE

The laboratory-level observations are linked to the conceptional level end-point by induction on the phenomena (*sensu* Thomas, 1995). Induction relies on the modeller's conception of how the various concepts are linked to one another, and how the concepts are linked to the material world. If A and B are phenomena at different organisational and conceptual levels, then the question "If the knowledge of the sate of A changes, will it impact on the sate of B" has to be repeated for all the phenomena under consideration. This implies that a system analytical model of the interactions be constructed based on the current insights on the system.

(a) As a first step a diagram as shown in Figure 2.2 might be generated where the direction of the arrows indicates the direction of influence. This also means that with equal validity a different conceptualisation will lead to a different model.

(b) The next step is to quantify the influence relationships. This would involve a) the quantitative or qualitative change in one state of one phenomenon as a function of the change in state of another phenomenon/phenomena, and b) the strength of that relationship.

INFERENCE MODEL STRUCTURE

Return now to the problem of estimating the likelihood of sustainability (or more precisely the unsustainability, which is defined largely at a conceptual level) based on current knowledge and observational data. It is necessary to link current understanding of ecosystem concepts to the stressors that are to be managed in such a way that finally the likelihood of ecosystem sustainability is expressed as a function of stressor characteristics.

The idea is to encapsulate system knowledge in a rule base expressing the relationships between phenomena (ρ). If ρ is combined with the site-specific evidence base (ε) in the form of a conjunctive combination, $\rho\wedge\varepsilon$ (where $\wedge$ indicates "conjunction"), then the outcome of this operation expresses the conclusion regarding the system status.

The rule base ρ can be rewritten in the canonical form to illustrate how it can be combined with the evidence ε in the two most often used forms of reasoning, the *modus ponens* and the *modus tollens* (DuBois and Prade, 1988).

Modus ponens:

Rule (ρ):	If V is A_1 then U is B	
Observation (ε):	<u>V is A_2</u>	[2.1]
Conclusion ($\rho\wedge\varepsilon$):	U is B'	

Modus tollens:

Rule (ρ):	If V is A then U is B1	
Observation (ε):	U is B2	[2.2]
Conclusion ($\rho \wedge \varepsilon$):	V is A'	

Step 1: Constructing an influence diagram

By repeatedly applying a *modus ponens* or *modus tollens* reasoning, a conclusion can be drawn regarding the truth of the antecedent.

From the postulates and the inference rule base in Section 2.4.1, a typical "fault tree" type of diagram can be constructed as shown in Figure 2.6. This is the basis of the phenomenological model.

Step 2: Quantifying the influence relationships

Applying this format (Eqs. [2.1] and [2.2]) to the postulates and the rule base in the appendix yields expressions like Eqs. [2.3] to [2.7] below.

Rule VIa :	$lc1 \wedge dc0 \rightarrow Cmps$	(η_6 true) [1]	
Observation:	lc1	(α true)	[2.3]
Observation:	dc0	(β true)	
Conclusion:	Cmps	(γ true)	

Rule Va:	$Cmps \rightarrow \neg Cmp$	(η_5 true)	
Observation:	Cmps	(γ true)	[2.4]
Conclusion:	$\neg$Cmp	(χ true)	

Sus: Sustainability is assured, *Res*: Resilience is assured, *Int* : Integrity is assured, *Div*: Biodiversity is intact, *Tpat*: Temporal stress/recovery patterns are undisturbed, *Cmp*: System composition is undisturbed, *Str*: System structure is undisturbed, *Fct*: System function is normal, *Tpats*: Temporal stress/ recovery patterns are in a state of stress, *Cmps*: System composition is under stress, *Strs*: System structure is under stress, *Fcts*: System function is under stress, *lxi0*: Minimally significant level of stressor *X* exists for integrity component *i*, *dxi0*: Minimally significant duration of exposure to stressor *X* exists for integrity component *i*, *dxi* : Long duration of exposure to stressor *X* exists for integrity component *i*, *lxi*: Intense exposure to stressor *X* exists for integrity component *i*, where $X \in$ {toxic substances (*T*), flow deficiency (*Q*), nutrient disruption (*N*), system driving variables disruption (*S*), physical habitat disruption (*H*)}, and $i \in$ {*Cmp* (*c*), *Fct* (*f*), *Str* (*s*), *Tpat* (*t*)}.

[1] $\rightarrow$ indicates logical implication and $\neg$ indicates "not" or logical negation

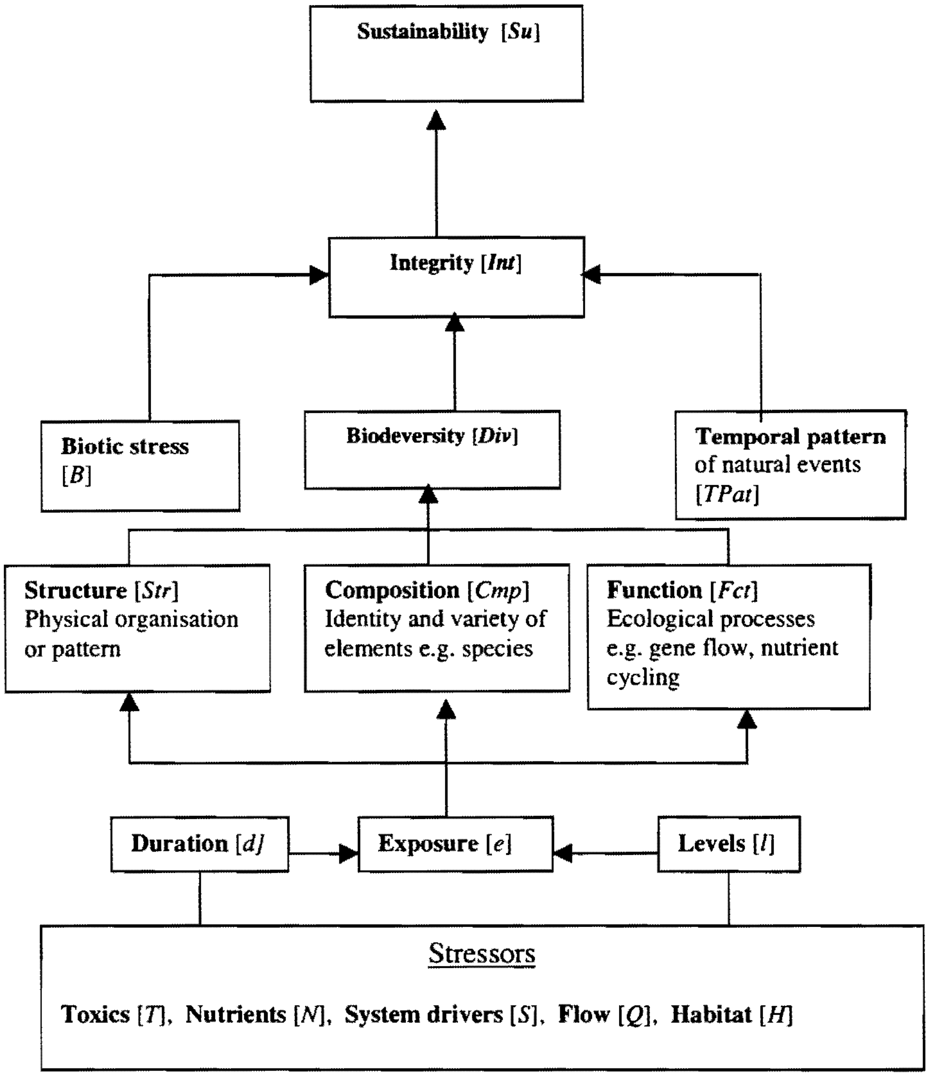


Figure 2.6. A systems model for ecological effect inference. The boxes indicate phenomena. The direction of the arrows shows the direction of influence. The blocks in Figure 2.6 must be read in conjunction with necessary characteristic, e.g. "Integrity" should be read: "The phenomenon of attainment of suitable integrity".

Rule IV:	$\neg$Cmp $\rightarrow$ $\neg$Div	(η_4 true)	
<u>Observation:</u>	<u>$\neg$Cmp</u>	<u>(χ true)</u>	[2.5]
Conclusion:	$\neg$Div	(δ true)	

Rule III:	$\neg$Div $\rightarrow$ $\neg$Int	(η_3 true)	
<u>Observation:</u>	<u>$\neg$Div</u>	<u>(δ true)</u>	[2.6]
Conclusion:	$\neg$Int	(ε true)	

Rules I and II:	$\neg$Int $\rightarrow$ $\neg$Sus	(η_2 true)	
<u>Observation:</u>	<u>$\neg$Int</u>	<u>(ε true)</u>	[2.7]
Conclusion:	$\neg$Sus	(τ true)	

Similar reasoning can be used for all the stressors and for other elements of biodiversity. Finally a conclusion can be reached as to the truth-value of the end-point: (e.g. "It is largely true that sustainability will be maintained" or " It 85% probable that the system will maintain > 95% sustainability").

2.4.4 DERIVING A RISK EXPRESSION FROM THE INFERENTIAL RULE BASE

Eqs. [2.3] to [2.7] express the inference of the system sustainability from the characteristics of the stressors occurrence. However, this inference is not yet a risk measure. It should be recalled that each of these inferences is the subject of an observer's conception induced onto perceptions of phenomena and that there is a measure of uncertainty in each inference. If it is supposed that the uncertainty can be described by a likelihood measure, Λ, that expresses an observer's (or a body of observers') confidence in the inference, then the measure of likelihood, $\Lambda(\neg Sus)$ is a risk measure. Each of the inferences can be represented by a conditional likelihood of the form: if A $\rightarrow$ B the uncertainty in the inference can be assessed by $\Lambda(A \mid B)$, i.e. the conditional likelihood of A given B. The exact form of the reduced likelihood depends on the measure Λ. Two types of likelihood measure are commonly used; each based on a different logic and each with its own calculus:

1) If the underlying **logic is crisp** (i.e. each proposition in the rule base is either true or false and nothing else, i.e. the values η_2 to $\eta_6 \in \{0, 1\}$ where 0 denotes "false" and 1 denotes "true") then results of probability theory are applicable and, consequently, Λ is then a **probability** measure and the results would belong to the domain of probabilistic risk assessment.
2) If the underlying **logic is fuzzy**, i.e. the values η_2 to $\eta_6 \in [0, 1]$, then the results of possibility theory are applicable and, consequently, Λ could be any one of a number of **possibility measures** each with a different interpretation and the risk will be possibilistic. Many of the phenomena (such as the existence of integrity) are essentially vague, and it is likely to benefit from a fuzzy approach.

Interpretation of the terms "risk", "probability" and "possibility" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997). The interpretation of likelihood is crucial to decision-making in data-poor ecological management situations.

2.4.5 SET-THEORETICAL DESCRIPTION OF THE INFERENCE PROBLEM

A set theoretical description is used to illustrate the point. Assume, for example, that the phenomenon "Ecosystem sustainability is lost" is used as an end-point. It is known that an infinite number of combinations of stressor states can result in this phenomenon. Assume that all the combinations of stressor states that correspond to the event: "sustainability is lost" are assembled in a set.

In defining the end-point phenomenon, the questions now arise: "At what point or combination of events can it be said that ecosystem sustainability is 'lost'? Is there a specific point at which it can be said that sustainability is lost? Or is there rather an increasing confidence in describing the system as being unsustainable?" The answers to these question can be summarised as in Table 2.3.

Table 2.3 The assessment of the state of the end-point phenomenon (loss of sustainability) and the state of lower level phenomena.

Case	**End-point phenomenon (set boundary)**	**Component phenomena (elements of the set)**	**Interpretation**
A	Crisp	Crisp	There exists a clearly defined set of threshold values that define a unique point representing system unsustainability.
B	Fuzzy	Crisp	Although the component events are clearly defined, the state corresponding to system collapse is vague.
C	Crisp	Fuzzy	The point of collapse is clearly defined but is not known how or when that state is reached.
D	Fuzzy	Fuzzy	Neither the point of collapse nor the threshold values are clearly defined.

The answers to these questions clearly lend different interpretations to the term risk since the likelihood that a parameter vector belongs to this set defines the risk.

If A and C are true it may still be that the parameters are subject to stochasticity. In this case risk is interpreted as the likelihood that a particular parameter vector of event states will belong to the set or not. Likelihood can be described in terms of probability theory, which requires a definable event to activate its precepts. In contrast to the frequentist view of probability, where probability is a limiting value of a series of repeated observations, the Bayesian view, where probability characterises the observer's sense of expectation, based perhaps on morphologically similar situations, can be used.

At the other end of the conceptual spectrum is the situation where B and D are true. The likelihood cannot be expressed in terms of the *probability* that a parameter vector belongs the set because the set and its elements are ill defined. The only recourse is to express likelihood in terms of fuzzy set theoretical likelihood measures such as possibility and necessity.

2.5 PROBABILISTIC FORMULATION OF THE END-POINT INFERENCE PROBLEM

In the literature referenced in this study, wherever risk is characterised quantitatively, the likelihood is expressed in terms of probability. Interpretation of the terms "risk" and "probability" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997) and particularly to decision-making in data-poor ecological management situations.

2.5.1 PROBABILITY THEOTERICAL APPROACH

Two approaches to a probabilistic expression of likelihood can be distinguished:

- The "frequentist" approach (Jaynes, 1996), sees probability as the limiting frequency of an occurrence over a large number of observations.
- In contrast, probability can be seen as a subjective expression (not necessarily dependent on repetitive observations) needed to project from the domain of uncertainty by the means of prevision to the domain of certainty. "Prevision, ... consists in considering, after careful reflection, all the possible alternatives, in order to distribute among them, in the way which will appear most appropriate, one's own expectations, one's own sensations of probability" (DeFinetti, 1990). With this view in mind, probability, and by association risk, could be seen as epistemic of the specific combination of situation and assessor.

Regulatory decision-making in the field of ecology is characterised by:

✓ A descriptive conceptual knowledge of ecosystems, often only supported by patchy observation.

✓ Observations of multiple replicates of experiments are often not available or simply impossible. The only recourse is then to expert prevision pertaining to a specific situation.

This is still in keeping with the principle of risk assessment. Predictive ecological risk is essentially an expectation of an effect, a prevision based on best available knowledge of the assessor's knowledge of and expertise in dealing with, what are as yet, unobserved events in a complex system. The calculated ecological risk values are therefore an expression of the assessor's expectation, taking into consideration the scientific information at hand.

In this section the expression of likelihood as probability is considered. (Note: Likelihood is not be confused with "likelihood" or "likelihood ratio" used in Bayesian statistics.) In expressing the uncertainty about the inferential expressions in the model, the use of probability theory was mentioned in respect of the use of binary or Boolean logic.

UPPER-LEVEL PHENOMENA

For those phenomena that are naturally concerned with levels of ecological organisation above that of population, the crucial inferences are Eqs. [A2.7] to [A2.9] in the Appendix

$\neg(Cmp \wedge Str \wedge Fct) = \neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div$ [A2.7]

$\neg(Div \wedge Tpat) = \neg Div \vee \neg Tpat \rightarrow \neg Int$ [A2.8]

$\neg Int \leftrightarrow \neg Sus$ [A2.9][2]

Each of the elements (Cmp, Str, Div, Sus, etc) refers to an end-point phenomenon that is considered relevant to a specific ERA or ERBM situation.

Given the uncertainty in both the arguments and the inferences, the probabilistic ecological risk would mean that Eqs. [A2.7 to [A2.9] need to solved by application of Eq.[2.8] which refers to generic events p and q and probabilities a and b (Dubois and Prade, 1988) to yield the set of equations [2.9].

$$\begin{array}{l} \mathrm{P}(\mathrm{p} \rightarrow \mathrm{q}) \geq a \\ \underline{\mathrm{P}(\mathrm{p}) \geq b} \\ \mathrm{P}(\mathrm{q}) \geq ab \end{array} \qquad [2.8]$$

$$\begin{array}{l} \mathrm{P}(\neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div) \geq \eta_3 \\ \underline{\mathrm{P}(\neg Cmp \vee \neg Str \vee \neg Fct) \geq \beta} \\ \mathrm{P}(\neg \mathrm{Div}) \geq \eta_3\beta \\ \\ \mathrm{P}(\neg Div \vee \neg Tpat \rightarrow \neg Int) \geq \eta_2 \\ \underline{\mathrm{P}(\neg \mathrm{Div} \vee \neg \mathrm{Tpat}) \geq \alpha} \\ \mathrm{P}(\neg Int) \geq \alpha\eta_2 \\ \\ \mathrm{P}(\neg Int \leftrightarrow \neg Sus) \geq \tau \\ \mathrm{P}\ (\neg Sus) \geq \tau\alpha\eta_2 \end{array} \qquad [2.9]$$

[2] $\leftrightarrow$ Denotes "if and only if" or logical equivalence.

If phenomenon p is considered logically equivalent to phenomenon q (i.e. $p \leftrightarrow q$) it is tantamount to asserting that one's knowledge of the uncertainty of the occurrence of p is no different from ones knowledge of the uncertainty of q and therefore $P(p) = P(q)$. However, the confidence in, or strength of the relationship (a in Eq. [2.8]) expressed as $P(p \leftrightarrow q)$ still needs to be assessed.

The probability of conjunction of phenomena in Eq. [2.9] may be difficult or impossible to assess. That would mean having knowledge of any of the endpoint phenomena occurring while the data at hand may only refer to the occurrence of phenomena in isolation. Consequently it is necessary to resolve the conjunction in terms of the probability of occurrence of individual end-point phenomena. The partitioning of a composite event probability into component event probabilities is accomplished by Eq. [2.10] (DeFinetti, 1990) where an event E is partitioned into n different logically independent events E_i where $i \in \{1,2,\ldots,n\}$. :to the conjunctions in Eq. [2.9] to the set Eq. [2.11].

$$P(E) = P\left(\bigcup_{i=1}^{n} P(E_i)\right) = \sum_i P(E_i) - \sum_{i \neq j} P(E_i E_j) + \sum_{i \neq j \neq k} P(E_i E_j E_k) - \ldots \pm P(E_1 \ldots E_n) \qquad [2.10]$$

Eq. [2.10] now contains terms that require the probabilities of conjunctions. These may be even less well known in an ecosystem context than the corresponding disjunctions. However, if one were to assume that the end-point phenomena are independent (i.e. that one's knowledge of the occurrence of one end-point in the conjunction is independent of one's knowledge of the occurrence of the other end-points), then the probability of the conjunction becomes the product of the individual phenomena probabilities.

Furthermore, analysis of Eq [2.10] (with the assumption of independence included) shows that Eq. [2.11] will always be true.

$$\max_i \{P(E_i)\} \leq P\left(\bigcup_i E_i\right) \leq \min_i \left\{\left(\sum_i P(E_i)\right), 1\right\} \qquad [2.11]$$

If the individual phenomena probabilities are known:

$P(\neg Cmp) = \beta_1$, $P(\neg Str) = \beta_2$, $P(\neg Fct) = \beta_3$ and $P(\neg Tpat) \geq \alpha_1$

Then

$\beta = \max\ \{\beta_1, \beta_2, \beta_3\}$

$\alpha = \max\ \{\alpha_1, \eta_3\beta\} = \max\{\alpha_1, \eta_3\beta_1, \eta_3\beta_2, \eta_3\beta_3\}$

$P(\neg Sus) \geq \tau\alpha\eta_2 = \max\{\tau\eta_2\alpha_1, \tau\eta_2\eta_3\beta_1, \tau\eta_2\eta_3\beta_2, \tau\eta_2\eta_3\beta_3\}$ [2.12]

Up to this point only higher-level phenomena had been addressed. A connection between higher and lower (laboratory-level) phenomena is proposed in Appendix 2.10.3. The combination of higher and lower level phenomena is shown diagrammatically in Figure 2.7.

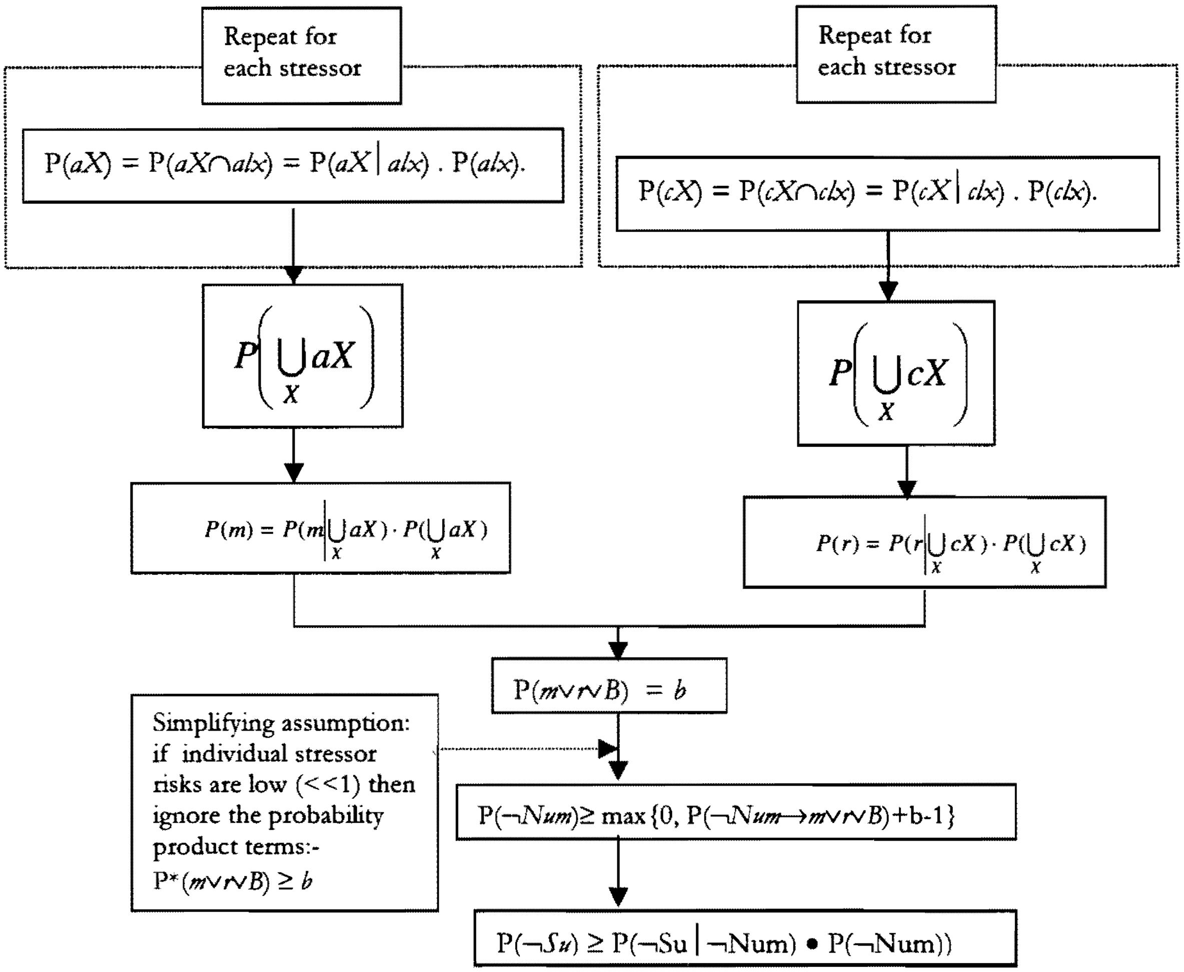


Figure 2.7 A diagrammatic representation of the process for estimating the confidence in high-level end-points from low-level (e.g. laboratory-level) end points.

2.6 POSSIBILISTIC FORMULATION OF THE END-POINT INFERENCE PROBLEM

2.6.1 BACKGROUND TO FUZZY APPROACH

The concept of fuzzy sets is commonly ascribed to the early work by Zadeh (1965). The essential difference between fuzzy and classic (crisp) sets lies in the definition of the sets. For crisp sets the universe of discourse is dichotomised into those events that belong to the set and those that do not (Klir and Folger, 1988), i.e. there must be a bijection between the sample space and the event space (Dubois and Prade, 1988). A probabilistic model is suitable for precise but dispersed information. In many real life complex situations this type of distinction is not that easy to make.

Each event is assigned a degree to which it is perceived to belong to the set under discussion (degree of membership μ).

2.6.2 THE RATIONALE FOR A FUZZY APPROACH TO RISK

Possibility theory (based on fuzzy set theory) (DuBois and Prade, 1988) may be better suited to the kind of situation where semi-quantitative expert opinion, such as in ecology, is the basis of the decision-making process. A fuzzy mathematical approach to ecological risk has been used (e.g. Ferson and Kuhn, 1993; Ferson, 1994) and possibility theory merits investigation as a total risk estimation tool.

Ecosystem characteristics

Some ecosystem characteristics could be interpreted at both a phenomenological and a mechanistic level. Concepts such as sustainability and resilience may be spatially and temporally scale dependent and the knowledge of the mechanisms underpinning these phenomena are vague (Costanza *et al.* 1993, De Leo and Levin, 1997). However, changes in the state of these phenomena are observable. As an example of the complexity of the mechanics related to such phenomena, is the natural variability and successional cycling in a system, which drives many of the ecosystem processes. If these are disrupted, a system may be produced that is structurally different to the original system. "Therefore, in managing ecosystems, the goal should not be to eliminate all forms of disturbance, but rather to maintain processes within limits or ranges of variation that may be considered natural, historic or acceptable" (De Leo and Levin, 1997).

Not only natural variability has to be accounted for in the management process, but also uncertainty and in some cases vagueness. Some definitions of ecosystem integrity; e.g. "the maintenance of the community structure and function characteristic of a particular locale or deemed satisfactory to society" (Cairns, 1977) or "the capability of supporting and maintaining a balanced, integrative, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley, 1981), although epistemic, is essentially vague and subjective. The system boundaries, the response to stressors and the stressors themselves may only be known qualitatively. The functional entities that best reflect the goals of ecosystem management may only be vaguely identifiable. Consequently, in dealing with ecological risk in the context of protective ecosystem management, it would be advantageous to use a paradigm that is adapted to address both uncertainty and vagueness such as possibility theory, which is based on the use of fuzzy logic.

Fuzziness in the inference model

The response inference model of (Appendix Eqs. [A2.1] to [A2.5d]), is essentially based on inference of form (*sensu* Thomas, 1995) rather than content. Eqs. [A2.4] to [A2.7d] and [A2.8a] to [A2.11] are expressions based on the formalisms of Aristotelian logic. If the assertion: A→B is made, this was essentially accepted as being true or false. In the probabilistic formulation of the inference model in Section 2.5, it was assumed that, due mostly to variability, there was a certain probability that this implication was either true or false. The only source of the uncertainty in this case was the variability in individual responses (stress) to stressors and the variability in exposure of the target entities to stressors. Consequently, the unique identification of both target entities and end-points for assessment was considered crucial.

However, if the definitions of sustainability, resilience and integrity, are considered, it becomes clear that it is not that easy to define target entities such as the ecosystem or what exactly is meant by "compromised sustainability", "loss of resilience", "compromised integrity", "corrupted composition", "abnormal system function", etc. There is an additional uncertainty imposed by vagueness in terminology that can only be eradicated by rigorous definition, which is unlikely to be mirrored in the precision and extent of the knowledge base or the definition of the system boundaries.

Moreover, it is likely that measures such as normality and integrity would be interval valued rather than single valued. All the assessments in the rule base may have to be made with reference to the condition of being intact or pristine. With an uncertain (fuzzy) knowledge base the assessment of *Fct* and *Int*, for example, would generally be of the type: "Largely normal" or "significantly impaired". However, the condition of being "undisturbed" is difficult to establish, but an observation about the system may to a greater or lesser degree be said to correspond to the condition of being "undisturbed". This means that both antecedents and consequents in Eqs [2.3 to 2.7] are fuzzy quantities. This places a suitable model for end-point projection in either Cases B or D of Table 2.3, but most likely in Case D.

A FUZZY INTERPRETATION OF THE ECOLOGICAL INFERENCE MODEL

In general, the rules on which the inferences are based are of the form "If $\tilde{X}$ is $\tilde{A}$ then $\tilde{Y}$ is $\tilde{B}$" where $\tilde{X}$, $\tilde{A}$, $\tilde{Y}$ and $\tilde{B}$ are generally vague. Recall that the propositions on which these rules in the Appendix were based refer to the pristine state. Rule I (See Appendix) could then be expressed alternatively as " The assurance of sustainability of the system takes its value from the (fuzzy) set of pristine values". It seems unlikely that the value of assurance of sustainability could have been given a specific value that would have been measurable and which could have been given a numerical value, since sustainability is merely a concept. At best the adequacy of the system sustainability could have been described as "very high" in a pristine system.

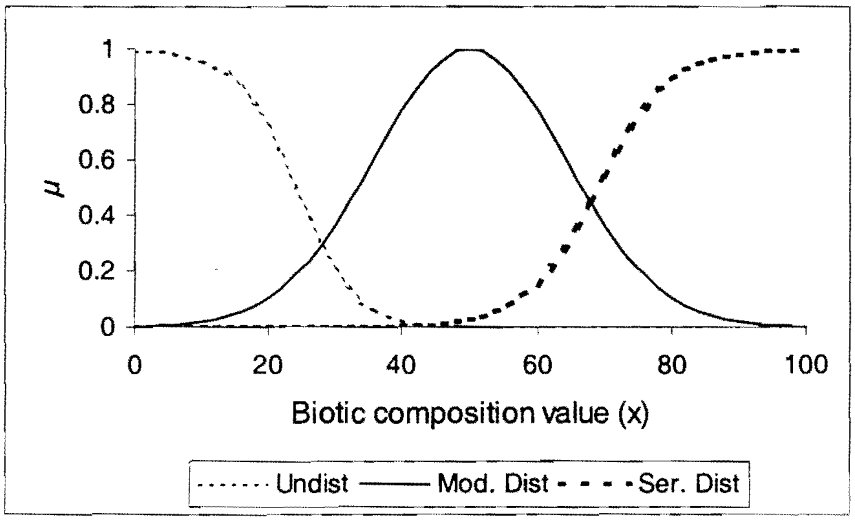


Figure 2.8. Example of a relationship between the value of a hypothetical biotic composition variable x and the degree to which it corresponds to the description "Undisturbed", "Moderately Disturbed" and "Seriously Disturbed". It is assumed that $x \in [0, 100]$ with 0 indicating a definitely seriously disturbed condition and 100 indicating a definitely undisturbed condition.

This degree of correspondence to the state of being pristine is expressed by the membership function μ of an observation x with respect to the set *Cmp* and is expressed as: $\mu_{Cmp}(x)$. In principle a variable x related to system biotic composition can be evaluated and a curve set up that relates the value of x to the degree with which it corresponds to the state of being undisturbed (i.e. $\mu_{Cmp}(x)$). The qualifier "undisturbed" might also be replaced with "mildly disturbed" or "seriously disturbed". This will give rise to series of curves as shown in Figure 2.8.

2.6.3 POSSIBILITY THEORETICAL APPROACH

Fuzzy logic is better geared to handle the domain of vague premises and conclusions and, consequently, the likelihood operator, Λ (Section 2.4.4), can best be replaced by the possibilistic counterparts from the domain of possibility theory.

Analogous to the relationship of probability theory to crisp set theory is the relationship of possibility theory to fuzzy set theory. One of the features of the application of fuzzy set theory and possibility theory is the ability to use non-numeric quantifiers in computing. It is inherently able to deal with both numeric and non-numeric data. Probability theory has no means to distinguish a state of certain knowledge that a system is stochastic and the state of knowledge uncertainty about a deterministic event. Possibility theory makes this distinction (DuBois and Prade, 1988).

If x takes its value over V, and y takes its value over U, and furthermore if V and U are normalised sets (i.e. sets where $\exists\, x \in V$ such that $\mu_V(x) = 1$), then the rule and observation and conclusion can be formulated in terms of possibility distributions or membership functions (DuBois and Prade, 1988) for *modus ponens* and *modus tollens* as Eqs. [2.13 and [2.14] respectively.

$\mu_{B'}(y) = \sup(\pi_{U|V}(x,y) * \pi_V(x)) = \sup[(\mu_{A1}(x)*\!\!\rightarrow\mu_B(y))*\mu_{A2}(x)]$ [2.13]

$\pi_A(x) = \mu_{A'}(x) = \sup[(\mu_A(x)*\!\!\rightarrow\mu_{B1}(y))*\mu_{B2}(y)]$ [2.14]

where the operators * and *→ are dependent in the implication used as defined in Table 2.3

The inferential problem can be solved by determining the truth-value of $(A_2 \wedge (A_1 \rightarrow B))$. The conjunction is represented by the t-norm (T): $B' = \sup[T(A_2, (A_1 \rightarrow B))]$, where sup indicates the supremum over all the values over which A_1 and A_2 are evaluated.

*Table 2.3. The form of the fuzzy operator * (t-norm), the corresponding t-conorm and the fuzzy implication operator (*→) (Klir and Folger, 1988)*

Logic	a*b (t-norm)	t-conorm	a*→b
Gödel	Min(a, b)	max(a, b)	= 1 if a ≤ b = b if a > b
Goguen	a · b	a + b – ab	= 1 if a = 0 = min(1, b/a) otherwise
Lukasiewicz	Max(0, a+b-1)	max(a+b, 1)	min(1, 1-a+b)

The approach to characterising the truth-values derives from the observation that each of the inferential rules can be expressed as a conditional likelihood describing the confidence the assessor has in the veracity of the rule. The rules can also be rewritten as possibility distributions:

Rule I and II: $\Pi(Sus \mid Int) = \eta_2$

Rule III: $\Pi(Int \mid Div \wedge Tpat) = \eta_3$

Rule IV: $\Pi(Div \mid Cmp \wedge Str \wedge Fct) = \eta_4$

Rule Va: $\Pi(\neg Cmps \mid Cmp) = \eta_5$

Rule VIa: $\Pi(Cmp \mid (l1c0 \wedge d1c) \vee (l1c \wedge d1c0) \vee (l2c0 \wedge d2c) \vee (l2c \wedge d2c0)\ldots) = \eta_6$

Applying Eq. [2.13] to the set of conditions above yield the fuzzy truth value for the end-point ¬Sus:

$\Pi(\neg S_{us}) = \sup\{T(\eta_2, \varepsilon), T(\eta_3, \delta), T(\eta_4, \chi), T(\eta_5, \gamma), T(\eta_6, \alpha, \beta)\}$, where T indicates a suitable t-norm.

If the min operator is chosen as the t-norm, then the possibility of unsustainability as an end-point is given by Eq. [2.15].

$$\Pi(\neg S_{us}) = \sup\{\min(\eta_2, \varepsilon), \min(\eta_3, \delta), \min(\eta_4, \chi), \min(\eta_5, \gamma), \min(\eta_6, \alpha, \beta)\} \qquad [2.15]$$

CHOICE OF AGGREGATION OPERATOR

A number of *t-norms* and *t-conorms* have been developed in multi-valued logic and which are used to express intersection and union of fuzzy sets respectively. The most commonly used of these are listed in Table 2.3 (DuBois and Prade, 1988, Kruse *et al.*, 1994). The choice of these *t-norms* and *t-conorms* is not an implicit part of the process but have they to be explicitly chosen. Klir and Yuan, (1995) lists a number of axioms which could be criteria for the selection of operators. Two of those which may be particularly applicable to this model (in addition to the one above) and which stems from a requirement that fuzzy logic should collapse to Aristotelian logic, are:

- The equivalence of $a\rightarrow(b\rightarrow x)$ and $b\rightarrow(a\rightarrow x)$
- $a\rightarrow b$ is true if and only if $a \leq b$, i.e. fuzzy implications are true if and only if the consequent is at least as true as the antecedent.

2.6.4 APPROACH DEPENDENT RISK INTERPRETATION

A comparison of the interpretation of risk in probabilistic and possibilistic terms is given in Table 2.4. Risk expressed in **probabilistic terms** implicitly has the interpretation that if a similar set of conditions such as stressor exposure and stressor effect is observed often enough, the probability component of the risk will express the **number of times the end-point will be expected to be observed**.

On the other hand, with the **possibilistic (fuzzy) expression** of risk, an observer's description of the endpoint phenomenon will always have a sense of uncertainty irrespective of how many times a similar set of stressor states is observed. In the fuzzy interpretation, the risk corresponds to the observed or predicted **state corresponding to the notion of the end-point**.

The difference in interpretation can affect the "proveability" of risk. A probabilistic risk expression raises the possibility that if enough instances of identical stress are observed, the end-point effect will be observed because the end-point is ontologically certain. In contrast, the fuzzy risk expression is the result of epistemic or systemic uncertainty. Even if the expected end-point is not observed, each result observed under stress *similar* (but not necessarily the same) to that being modelled, will add to the evidence base, which either supports or rejects the risk characterisation.

Table 2.4. A comparison of the interpretation of risk in probabilistic and possibilistic terms.

Aspect of risk assessment	Probabilistic	Possibilistic
Basis	Probability theory	Possibility theory/ fuzzy logic
End-point type	Crisp events	Vague / fuzzy events
Exposure assessment	Probability density distribution	Possibility or necessity distribution
Effect assessment	Cumulative probability of effect conditional on exposure	Cumulative possibility/ necessity of effect conditional on exposure OR Implication operator OR rule-base
Likelihood characterisation	Product	Implication related t-norm/ t-conorm operator (e.g. min/ max)
Stressor likelihood integration	Sum-product rule	Max - min operators

2.7 DERIVING AND INFORMING STRESSOR RESPONSE RELATIONSHIPS

2.7.1 INTRODUCTION

In Figure 2.1 it is shown that the only parallel tasks in ERA are effect assessment and exposure assessment. Of these exposure assessment has the advantage of a number of models being available for predictive exposure assessment.. For substances, models such as WARNB, MCARLO and SIMCAT could be used with stochastic inputs to calculate effluent criteria and TOMCAT and QUAL2E in addition to the others can be use to estimate in-stream substance concentrations (Ragas, *et al.*, 1997). A number of flow models also exist (e.g. the Pitman model commonly used in South Africa (Pitman, 1973)). At present it is not known whether any models exist to predict habitat degradation as a stressor and it appears likely that habitat degradation will remain to be assessed *in situ*. Therefore, a combination of observation and modelling can be used to estimate the stressor exposure likelihood.

The other component in the risk estimate, effect likelihood, was characterised as the likelihood of effect conditional on the exposure as represented by the stressor-response-relationship (SRR). In its simplest form an SRR could be characterised by a lower and an upper acceptability limit as illustrated in Figure 2.9.

The **minimum characteristics** of an SRR for effect likelihood are:

1. It must express the relationship between a level of stressor and the level of occurrence of the end-point.

2. It should be able to resolve the stressor-levels where there is no expectation of end-point response and complete expectation of end-point response.

3. In its simplest form it could be a discontinuous stepped function as shown in Figure 2.9, but it could also be a smooth s-shaped curve. The form shown in Figure 2.9 indicates an increasing expectation as the stressor metric increases. The acceptance limits need not represent discontinuities but may be interpreted as selected percentiles of a suitable cumulative distribution curve or some other suitable function as long as it reflects the **present state of knowledge**. A SRR could also be in the form of a rule base.

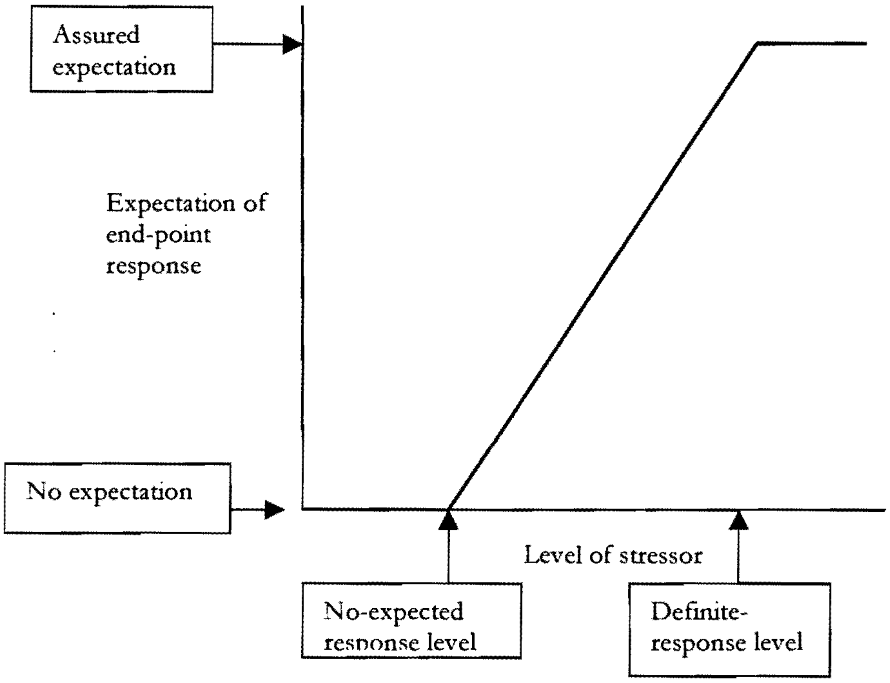


Figure 2.9 An illustration of the parameters needed to construct an SRR. The upper and lower expectation limits are stressor levels corresponding to unacceptable and acceptable levels of expectation of effect.

4. It must be **monotonic**, although it need not be strictly monotonic. That is, any given effect expectation should map to only one point or contiguous interval in the stressor level domain. A stressor that has a similar effect at very high and at very low levels should be modelled as two separate stressors. The reason for the monotonicity is to preserve consonance between the effect and stressor.

INFORMING TOXIC SUBSTANCE SRR'S

Ecotoxicological methods used in the bio-assessment of toxic substances are a solid basis for developing SRR's. The data as derived from toxicity tests serve as the basis for hazard assessment. Two common assumptions when applying these data in hazard assessment are (a) that exposures are temporally invariant and (b) that individual based tests apply directly at higher levels of organisation.

Effect data for toxic substances exist mostly at the *individual organism* level and, to a lesser extent, at the population level, while effect data for the other stressors exist largely t the population and community level. However, more realistic risk assessment is still hampered by a lack of knowledge of conditional probability of effect at higher levels of organisation. As a simplification, it is often assumed that an impact at the lower level of organisation (where the data exist) necessarily implies an impact at the higher level of organisation. Consequently, the risk predicted at the lower level of organisation is at least as great as that predicted at the higher level of organisation since the probability of a logical consequent cannot be greater than that of the antecedent. Although this is a reasonable starting point, if all the interactions have not been accounted for and the conditional probabilities evaluated, this assumption could be seriously in error. As a result, the calculation above, and indeed any risk assessment based on such a premise, could be seriously in error.

The assessment of the parameters in the temporally invariant case derives directly from ecotoxicological assessment. The higher the level of organisation represented in the test the better. Some notes on the use of population level projections from individual level assessments are made in 2.4.3. Temporally variable stressor levels are more realistically found in real stream quality management situations and these present a greater challenge. Some notes are appended on the estimation of probable mortality from temporally variable concentrations.

A brief overview of some of the issues involved in toxicity bio-assessment as the basis for toxicity SRR's appear in the Appendix 2.11. From this discussion it is clear that:

1. Since it is impossible to define a "most sensitive species" the estimation of a protection level is based statistical models. This implies the selection of toxicity test species should be as extensive as possible so that a suitable database can be generated for the statistical models.

2. While the bulk of toxicity data is generated at the individual organism level, this is generally not the best level for data on which to base ecosystem-level decisions. However, methods do exist to project from the individual organism-level to higher levels. These would include the methods referred to in Table 2.5.

3. Its is of particular importance to incorporate the effect of time variable toxic substance levels. Data on bioconcentration could be used very effectively in combination with pharmacokinetic models to estimate response expectation.

4. The interpretation and application of mixture toxicity data needs to be developed further in order to improve SRR quality.

Table 2.5 Some common methods for extrapolation of effects

Type	Extrapolation/ Projection	Form	Rationale	Reference
Bio-assessments	Stressor magnitude (e.g. concentration) to species level effect.	Concentration-response functions	Concentration proportional to receptor dose	e.g. Hathway, 1984
Response regression	From lower to next higher taxonomic level	Regression equations, projection matrices	Species representative of its taxon	Suter , 1993; Caswell, 1989; Suter, 1993; Caswell, 1996
Dose scaling	Across species	Allometric equations	Physiological functions proportional to physical characteristics (e.g. body mass, volume etc.)	Kenaga, 1978; Crouch, 1983; Chappell, 1992; Suter, 1993
Diet extrapolation	Across different trophic groups	Qualitative categories of susceptibility	Adaptation to common diet	Mullin, *et al.*, 1982; Suter, 1993
Guild extrapolation	Across different guilds	Qualitative similarities	Common diet and environment and similar behaviour within guilds	Cummins, 1974; Severinghaus, 1981

INFORMING FLOW AND HABITAT STRESSOR SRR'S

In contrast to toxic SRR's, the SRR's for flow and habitat stress is more likely to be derived from field observations with interpretation by experts in the field.

However, much work is being done from which flow-related stress and flow-related stressor-response information can be drawn (e.g. King and Louw, 1998; Hughes and Münster, 1999) and some experimental and or observational data exist from which the possibility of effect can be inferred (e.g. Chessman, *et al.*, 1987; Quinn, *et al.*, 1992; Cooper, 1993; Roux and Thirion, 1993; Thirion, 1993). It appears that much more research is needed to assess effects at *ecosystem* level.

An important feature of risk-based management is the feedback loop between the field bio-monitoring and the problem formulation and risk characterisation steps in risk assessment. Risk in itself cannot be proved to be correct or incorrect, but a formal methodology to adapt the process, will ensure dynamic, scientifically defensible risk management in a catchment.

From the discussion of an approach to derive habitat and flow SRR's it is clear that:

1. There is a dearth of information on habitat and flow stress and there is nowhere near the amount of controlled experimental data on which to base the SRR's compared to toxic SRR's. The use of a fuzzy expert system may in many cases be the only type of SRR available.

2. A fuzzy relationship of the form E = R o A may be used, where E is an effect, o is a suitable implication operator and A is a stimulus. R is the SRR for the stressor and would likely be in the form of a matrix.

3. In order to formulate R, there must exist a training set of stimuli and responses. Once R has been formulated it is applied in conjunction with observed or predicted stimuli to predict response expectation.

4. In the case of flow and habitat response, it is particularly necessary to develop the methodology to update R by using data from field observations. This can be done by the use of the Dempster-Schafer theory (DuBois and Prade, 1988). A considerable volume of work has been done on belief functions and their updating by Dempster-Schafer as well as other updating algorithms (Smets, 1981; 1991a,b; 1993; 1994).

2.9 CONCLUSIONS

The two major problems in applying risk methodology in ERBM relates to the effect assessment phase. This phase requires the formulation of a SRR, which must express the relationship between the stressor level and the expectation of the end-point effect. With regard to SRR's the two most obvious problems are: (a) the problem of estimating the risk at higher level end-points when only data at lower level end-points are available because the end-points are incompatible, and (b) informing the SRR.

The theoretical considerations presented in this chapter indicates that:

- Both uncertainty and variability are likely to be important in ERA and ERBM. There is clearly a need to ensure that risks are assessed at the correct organisational level and consequently there is a need to project the risk estimated at a lower organisational level to a higher organisational level. The uncertainty around end-point projection can be addressed by and phenomenological end-point projection model.
- The likelihood of ecological effect can be expressed either in probabilistic or possibilistic terms. The interpretations are compared in Table 2.4.
- A comparison of the form of Eqs. [2.12] and [2.15] shows that the probabilistic formulation will most likely yield the lower limit of expectation of the end-point while the possibilistic formulation will most likely yield the upper limit of expectation. Which one of the two is used will depend on the purpose of the risk assessment.

- Methods do exist to inform SRR's. Toxic SRR's can be based on the toxicity assessments. In this case it is particularly necessary that the risk end-points need to checked carefully. Other stressors, such as flow and habitat degradation, would more likely benefit from fuzzy expert system formulation of the SRR problem. In all cases, but especially in the case of flow and habitat stress, is it necessary to update the SRR from field observations. The challenge to risk management of multiple stressors will be the formulation of expert systems that are able to tap the ecological knowledge of the effect of stressors at higher levels of ecological organisation and express it in a form that can be used in ecological effect assessments. The assessment of the likelihood terms in the model is not a simple task.

- The choice of basis on which ecological effect likelihood is based should correspond to the characteristics of the end-point and nature of the data available. For crisp, well-defined events, which are uncertain in occurrence, a probabilistic formulation is well suited. If the end-point or the data is subject to epistemic uncertainty, then fuzzy logic and a possibilistic formulation is indicated.

CHAPTER 3

MODELLING THE DIVERSE STRESSOR PROBLEM

All models are wrong, but some are useful – George Box (1979)

3.1 SUMMARY

This Chapter deals with the problem of estimating the aggregate risk of a number of diverse stressors (referred to as the Diverse Stressor Problem).

There did not appear to be any formal mathematical formulation of ERA except for the Kelly-Roy-Harrison formulation. This formulation could be shown to be a special case of the probabilistic conjunction of stressor effect and stressor occurrence.

The aggregate risk of diverse stressors is modelled as the disjunctive occurrence of effects due to the different stressors. Both probabilistic and possibilistic formulations of this model were made and tested in hypothetical cases. These tests showed that the probabilistic formulation had more strenuous requirements regarding end-point definition and SRR input compared to the possibilistic formulation, but it is more likely to be applicable in law-enforcement. The fuzzy (possibilistic) formulation was more easily adapted to imprecise ecological data.

3.2 ESTIMATING THE AGGREGATE RISK OF DIVERSE STRESSORS: THE DIVERSE STRESSOR PROBLEM

3.2.1 THE KELLY-ROY-HARRISON EXPRESSION

Although the use of conditional probability (and other expressions of likelihood) is well known in risk assessment generally, it has not been obvious in literature on ERA. Kelly and Roy-Harrison (1998) note that mathematical formulation of ERA appears to be pointedly avoided for fear of misuse or misinterpretation. Nevertheless, they formulate risk (R) as a function of an adverse effect (E), the consequence of an adverse effect (C(E)) and the likelihood of adverse effect (Λ(E))which is expressed as a function of exposure (P) and the existence of a stressor (S) such that for *k* severity levels, *i* stressor levels and *j* exposure levels:

$$R = \sum_{k} C_k(E_k) \cdot \sum_{i} \sum_{j} \Lambda(E_k \mid P_{ji} \wedge S_i) \cdot \Lambda(P_{ji} \mid S_i) \cdot \Lambda(S_i) \qquad [3.1]$$

With regard to the Kelly-Roy-Harrison formulation (Eq. [3.1]) it should be noted that:

1. It makes provision for the situation where a stressor is given while the various consequences needs to be explored and quantified. In this study the focus is on the situation where the end-point is given (encapsulated in an ecosystem level phenomenon, e.g. loss of sustainability). This means that the consequences are discounted in the end-point and all that is left to determine is the likelihood of adverse effect. Furthermore, because ERBM focuses on management for a predetermined effect and its probability, both 'consequences' and 'adverse effect' (i.e. C(E)) is fixed by the regulatory requirements. Consequently, Eq. [3.1] practically reduces to Eq. [3.2].

$$R = \sum_{k} \sum_{i} \sum_{j} \Lambda(E_k \mid P_{ji} \wedge S_i) \cdot \Lambda(P_{ji} \mid S_i) \cdot \Lambda(S_i) \qquad [3.2]$$

2. Eq. [3.2] makes a distinction between stressor occurrence and exposure. In environmental assessment of the effect of chemicals, this is fundamentally correct because a stressor introduced into the environment may contact an organism by various routes simultaneously with each route contributing differently to the overall risk. In aquatic environments there may probably fewer routes of exposure and some are more likely to dominate. **In the short term**, direct intake of water is likely to dominate, while on the longer term indirect exposure may also contribute. In the view of Kelly and Roy-Harrison (*op. cit.*), for human and ecological risk assessment, $\Lambda(S)=1$. In other words, the stressor definitely occurs, it is only the exposure that may differ. For the purpose of this study, where for some stressors effect does not depend on uptake but on overall stress, it is assumed that occurrence and exposure are equivalent. It should be borne in mind that for chemicals (and particularly toxics) this assumption does not necessarily hold. For the purpose of this study Eq. [3.2] reduces to [3.3].

$$R = \sum_{k} \sum_{i} \Lambda(E_k \mid S_i) \cdot \Lambda(S_i) \qquad [3.3]$$

3. Eq. [3.3] still contains the summation over *k* severity levels of adverse effect and *i* stressor levels. Probability is expressed as probability density and consequently Eq. [3.3] is an expression of the area overlap between effect and exposure distributions. This stands in stark contrast to the calculation of risk by the quotient method (See Risk Characterisation Phase in Section 2.2.2) where two concentrations or stressor levels are compared. Eq. [3.3] is a more general form of risk expression.

4. Eq.[3.3] contains an expression of a SRR and a stressor occurrence expectation. This is in fact a special case of the probabilistic expression of the ***modus ponens*** inference (Eq. [2.1]) where the rule $S_i \rightarrow E_k$ and the observation S_i are combined (i.e. $R = \Lambda(S_i \rightarrow E_k \wedge S_i)$). This expression is analogous to the combination of the inferences in Eq. [2.9]. A more general expression that does not prescribe the way in which likelihood is to be expressed needs to be derived.

The other major problem still remains: how to estimate the aggregate risk when a number of different stressors occur.

3.2.2 CONJUNCTION-DISJUNCTION EXPRESSION

From the theoretical considerations in Chapter 2 it was established that a risk only occurs when (a) a stressor exists AND (b) the stressor (by definition) has an effect on some target entity in the ecosystem. Therefore, if the stressor existence is designated by S and the effect of the stressor is designated by E then a risk only exists when $(E \wedge S)$ is true. More precisely the risk is the likelihood that $(E \wedge S)$ is true: $R = \Lambda(E \wedge S)$.

The effect E is here a generalised expression of the observation that a stressor of the same type as S has an effect. This effect generally occurs over stressor set Y. However, risk is assessed for a specific situation, where particular values of Y, namely the set S will be found (i.e. $S \in X$). So risk for stressor X is the properly expressed as Eq. [3.4]

$$R_X = \Lambda((E_X \mid X) \wedge S) \qquad [3.4]$$

If likelihood is expressed in terms of probability then Eq. [3.4] becomes Eq. [3.5] while if it is expressed as possibility then it becomes Eq. [3.6]

$$R_X = P((E_X \mid X) \wedge S) \qquad [3.5]$$
$$R_X = \Pi((E_X \mid X) \wedge S) \qquad [3.6]$$

The effect E could, in the present context, be the occurrence of an event such as "loss of sustainability". Each stressor acting on an ecosystem may result in E either on its own or in conjunction with other stressors. So each stressor produces an individual risk of effect E. If stressors X, Y, Z… are present in the system and they occur on a site-specific basis as S, T, U, …, then the risk R of E due to either X OR Y OR Z OR … will be given by Eq.[3.7].

$$R = \Lambda\{((E_X \mid X) \wedge S) \vee ((E_Y \mid Y) \wedge T) \vee ((E_Z \mid Z) \wedge U) \vee \ldots\} \qquad [3.7]$$

R is an expression of the aggregate risk and is assessed in a manner similar to Eqs. [3.5] or [3.6]. Each of these individual stressor risks can be estimated by ERA. In order to assess the expectation of all the stressors acting at the same time, the individual stressor ERA outcomes need to be convoluted. There are several mathematical operators that can be used to convolute stressor risk to reflect the total risk, including: maximum, sum and conjunction. The specific operators will depend on whether a probabilistic or possibilistic formulation is used. These will be investigated in section 3.3 and 3.4 respectively. The event E will, in the rest of the Chapter, be partitioned into events that relate to the various types of anthropogenic stress, such as toxicity (t), flow regime disturbances (q) and habitat degradation (h).

3.3 PROBABILISTIC AGGREGATE OF DIVERSE STRESSOR RISK

3.3.1 BACKGROUND

In a probabilistic expression of the aggregate risk consider the event E in an ecosystem subject to n different stressors. Each stressor i will give rise to E_i. The combined probability of effect (in set theoretical terms) is given by (DeFinetti, 1990):

$$P(E) = P\left(\bigcup_{i=1}^{n} E_i\right) = \sum_i P(E_i) - \sum_{i,j} P(E_i E_j) + \sum_{i,j,h} P(E_i E_j E_h) - \ldots \pm P(E_1 E_2 \ldots E_n) \quad [3.8]$$

If E_t , E_q and E_h are all logically independent, then probability of the conjunction of individual ecological effects reduces to the product of the individual effect probabilities, and hence the application of Eq. [3.8] to Eq. [3.7] yields Eq. [3.9]:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)] \quad [3.9]$$

3.3.2 SYNERGISM OR ANATAGONISM AMONG STRESSORS

$P(\varepsilon_x | x)$ is defined as the probability of an end-point ε given the event that stressor X is present at level x. Furthermore, the effects ε_x may not be functions of one stressor only. It may be necessary to partition the event "existence of stressor X" into events that signify the occurrence of stressors that collectively manifest as stressor X: i.e. X is partitioned into occurrence of stressors (X_1 , X_2 , ...X_n), where there are n stressors that make up the class of stressor X. Due to interactions among stressors, it may be necessary to evaluate $P(\varepsilon_x|X)$ where all n different stressors are present at the same time. Most often this will not be possible experimentally (except perhaps in the case of toxic stress), so that simplifying assumptions will have to be made. However if stressor occurrence events X_j are logically independent then this reduces to Eq. [3.10] (DeFinetti, 1990).

$$P(\varepsilon_x|X) = \sum_j \left(P(X_j) \cdot P(\varepsilon_x|X_j)\right) \quad [3.10]$$

It might be, that although the stressor occurrences X_i and X_j are independent, the effect ε is dependent on the co-occurrence of X_i and X_j . This might be due to some mechanistic interdependence such as synergism or antagonism in which case the occurrence of (X_i,X_j) might manifest as a new stressor Y. In this case $P(\varepsilon|X_i X_j)$ would be given by: $P(\varepsilon_Y|Y) = P(\varepsilon, Y) / P(Y)$. Therefore, $P(\varepsilon,X_i,X_j) = P(X_i)P(X_j)P(\varepsilon| Y)$, where the value for $P(\varepsilon| Y)$ has to be evaluated experimentally. However, cases of true synergism among toxics, for example, are reported to be rare (Calamari and Vighi, 1992). The occurrence of synergism among other stressors may be possible.

3.3.3 A HYPOTHETICAL CASE STUDY

A hypothetical case study to illustrate an application of the above is given in Part 3, Paper 2.

3.4 POSSIBILISTIC AGGREGATE OF DIVERSE STRESSOR RISK

3.4.1 THEORETICAL BACKGROUND

The point of departure in formulating of aggregate ecological risk is Eq. [3.7]. Rewriting Eq. [3.7] for the three-stressor assumption yields Eq. [3.11]

$$R = \Lambda\{((E_Q \mid Q) \wedge q) \vee ((E_H \mid H) \wedge h) \vee ((E_T \mid T) \wedge t)\} \qquad [3.11]$$

The possibilistic approach to the ecological risk problem is formulated as the disjunction of the ecological risk rule base with predicted or observed stressor data. The risk rule is captured in the conditional likelihood. E is defined by the NWA as being "loss of sustainability" or $\neg Sus$. Each of the disjunctive terms in t Eq. [3.11] can be written in the form:

Rule:	X is A $\rightarrow \neg$ SusY is B	
Observation:	<u>X is A′</u>.	[3.12]
Conclusion:	$\neg$Sus is B′	

Each premise contains a characteristic ("sustainability") and an evaluation ("loss of"). In the case where the propositions in the premise can only be true or false (i.e. the application of "crisp" logic), the uncertainty is expressed in terms of probabilities.

The evaluation of the propositions in the case of most ecosystems is almost necessarily vague, epistemic of an observer in a situation and possibly phenomenological. In general, probabilities cannot be used to evaluate the likelihood of effect. In order to apply the well-established probability calculus to the estimation, the evaluations are given a numeric value so that Aristotelian logic applies. For example, if the evaluation "maintained" is replaced by "80% maintained" then the outcome of an assessment can be true or false in principle. This, however, requires either considerable ecosystem specific knowledge, or, simply assumption of a value as a norm. The nature of ecological assessments is often more amenable to vague assessments of these values such as: "high", "moderate" etc., which corresponds to typical fuzzy sets. So, the expressions A and B in [3.12] are fuzzy sets. Consequently, if t is a specific response to stimulus s, then Eq. [3.12] can be solved by (DuBois and Prade, 1988):

$$\mu_{B'}(t) = \sup_{s \in S}(\mu_A(s) * \rightarrow \mu_B(t)) * \mu_{A'}(s) \qquad [3.13]$$

where * is a suitable t-norm and *→ is the corresponding implication operator which could be replaced by the conditional possibility distribution $\pi_{Y|X}(s,t)$ if the sets are normalised.

In this study the evaluation was performed for four fuzzy sets so that A, B ∈ { Negligible, Low, Moderate, High}. For example [3.11] can be expressed as "IF effect of stressor 1 IS Negligible OR effect of stressor 2 IS Negligible OR… THEN NOT (Sustainability) IS Negligible"

For each stressor, $Poss(E_i)$ and $Nec(E_i)$ can be calculated (DuBois and Prade, 1988; Kruse, *et al.*, 1994):

$$Poss(E_1 \vee E_2 \vee E_3 \ldots) = \max\{Poss(E_1), Poss(E_2), Poss(E_3)\ldots\} \text{ and}$$
$$Nec(E_1 \vee E_2 \vee E_3 \ldots) \geq \max\{Nec(E_1), Nec(E_2), Nec(E_3) \ldots\} \qquad [3.14]$$

A more complete expression of the risk inference in terms of a conditional possibility or necessity measure (DuBois and Prade, 1988) is:

$$\begin{array}{l} Poss\,(X \mid E_X) \geq a' \\ Poss\,(E_X \mid X) \geq a \\ \underline{Poss\,(X) \in [b, b']} \\ Poss\,(E_X) \in [a*b, a'*\rightarrow b'] \end{array} \qquad [3.15]$$

$$\begin{array}{l} Nec(X \mid E_X) \geq a \\ Nec(E_X \mid X) \geq a' \\ Nec(X) \in [b, b'] \\ Nec(E_X) \in [\min(a, b), (1 \text{ if } a' \leq b' \text{ or } b' \text{ if } a' > b')] \end{array} \qquad [3.16]$$

The possibility and necessity measure are interpreted to mean the extent to which a fuzzy set may possibly correspond to a given description and the extent to which a fuzzy set may correspond to the complement of the fuzzy set respectively. For the probability measure, P, of set E_X, it is always true that $Nec(E_X) \leq P(E_X) \leq Poss(E_X)$. Consequently, it is possible to estimate the upper and lower limits for the possibilistic risk to the ecological sustainability from a knowledge of the possibility and necessity of the stressor levels which can be calculated from the possibility distributions of the stressors, the stressor response and some knowledge of the stressor impact structural biodiversity inference.

3.4.2 HYPOTHETICAL CASE STUDY

A hypothetical case study is described in Part 3, Paper 3.

3.5 INDEPENDENCE OF PHENOMENA

In the foregoing, the assumption of independence of phenomena featured strongly. One of the strongest objections to Jooste (2000) had been the assumption of independence among stressor phenomena. It was pointed out that it is well known that some substances act synergistically even though true synergism is reportedly quite rare. Furthermore, even among heterogeneous stressors it is quite conceivable that when two stressors occur together (e.g. flow insufficiency and toxic substances) that the stress caused by the one exacerbates the stress caused by the other, and although there is no true synergism, the effect would be qualitatively similar.

This objection appears to be due to the "Mind Projection Fallacy" (Jaynes, 1996) at work in risk assessment. It should be remembered that risk, although often expressed as a probability, is in fact a descriptor of the assessor's *state of knowledge*, assigned to a *phenomenon*. While it may incorporate knowledge of the mechanistic detail, once the descriptor for a particular set of stressor values is assigned, it loses that detail.

Consider a multiple stressor problem as follows: Assume that the phenomenon: {Unsustainability is caused by stressor **x** with value x } is indicated by X. Assume that stressor **y** with value y resulting in stress Y occurs simultaneously. It is important to note that a distinction is made between the *phenomenon* and the *mechanism* by which this phenomenon came about. For the risk assessment of X it would be important to know by which different mechanisms the phenomenon X was reached. If , for example, a probabilistic risk of X is considered then the risk would be given by $P(X|x \wedge y)$. This can be recognised as a Bayesian posterior distribution, which is the left-hand side of Eq. [3.17].

$$P(X \mid x \wedge y) = P(X \mid x) \cdot \frac{P(y \mid X \wedge x)}{P(y \mid x)} \qquad [3.17]$$

In general, the question should be asked in risk assessment whether there exists any knowledge of the likelihood ratio (i.e. the second term on the right hand side of the Bayes equation). The prior probability must, by definition, exist since that is the rationale for doing a risk assessment.

An assessment of the likelihood ratio begs the question of whether the existence of stressor value could have been inferred from a knowledge of the existence of stress X and the co-occurrence of stressor values x and y. In general it might be suspected that such a synergism exists, but proof is often lacking. If there is mechanistic reason to believe that y will potentiate (or exacerbate) the effect of x, then an assessment of the likelihood ratio can in principle be done. If no evidence exists, then the posterior probability equals the prior probability and the risk pertaining to the co-occurrence of the two stressors is no different from the risk of induced by x, i.e. the likelihood ratio is 1. However, if the likelihood ratio differs from 1 then the risk pertaining to the phenomenon X is given by the posterior distribution. The stressor values and their interaction have now been discounted in the risk calculation. Consequently, the risk of X for any given set of x and y will be independent of risk of Y. Therefore, it could be said that the risk of the phenomena X and Y are logically independent. So, although some causal dependence may exist, the risk of the phenomena may be logically independent. It seems particularly prudent in ecological risk assessment to be wary of the "Mind Projection Fallacy" (see below)

Jaynes (1996 *p* 406) describes the difference between causal and logical independence as follows: "Two events may in fact be causally dependent (i.e. one influences the other); but for the scientist who has not yet discovered this, the probabilities representing his state of knowledge – which determine the only inference he is able to make – might be independent. On the other hand, two events might be causally independent in the sense that neither exerts any causal influence on the other [...] yet we perceive a logical connection between them, so that new information about the one changes our state of knowledge of the other. Then for us their probabilities are not independent." He described this confusion between reality and a state of knowledge about reality as the "Mind Projection Fallacy".

3.6 AGGREGATION MODEL SUMMARY

The aggregation of the risk of diverse stressors make of the logical disjunction of individual stressor risk.

$$R = \Lambda\{((E_X \mid X) \wedge S) \vee ((E_Y \mid Y) \wedge T) \vee ((E_Z \mid Z) \wedge U) \vee \ldots\}$$

In probabilistic terms this model becomes:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)]$$

In possibilistic terms this model becomes:

$$Poss(E_t \vee E_q \vee E_h) = \max\{Poss(E_t), Poss(E_q), Poss(E_h)\} \text{ and}$$

$$Nec(E_t \vee E_q \vee E_h) \geq \max\{Nec(E_t), Nec(E_q), Nec(E_h)\}$$

The individual stressor risks are calculated from a SRR and a likelihood of stressor occurrence.

In probabilistic terms:

$$P(E_X) = P(\varepsilon_x \mid x) \,.\, P(x)$$

In possibilistic terms:

For a fuzzy descriptive set A or A′of stressor X and fuzzy descriptive set B or B′ of response Y:

$$\mu_{B'}(t) = \sup_{s \in S}(\mu_A(s)^* \rightarrow \mu_B(t)) * \mu_{A'}(s)$$

Poss (B′) = max { $\mu B'_i(t_i)$} over all stressors *i*

A comparison between the probabilistic and possibilistic formulation in Table 3.5 below shows that, at least in the short term, the fuzzy formulation might be more appropriate, although the regulatory requirement might motivate for clarifying the knowledge-base to allow for the use of the probabilistic formulation.

Table 3.5. A comparison between the probabilistic and possibilistic formulations of the diverse stressor problem.

Component	**Probabilistic**	**Possiblistic**
End-point	Crisp definition	Fuzzy or crisp definition
SRR-type	Unique	Unique or fuzzy
SRR data requirement	Extensive	Limited, expert system
Adaptability to diverse ecological stressors	Low (data limitations)	High
Applicability of results to law-enforcement	Well adapted	Difficult

CHAPTER 4

MODELLING THE DIVERSE-STRESSOR-MULTIPLE-SOURCE PROBLEM

4.1 SUMMARY

This Chapter deals with the solution to the diverse-stressor-multiple-source (DSMS) problem in the context of ecological risk-based management (ERBM). The effect disjunction solution to the diverse-source problem of Chapter 3 is used as the basis for solving the DSMS problem. ERBM requires that stressor profiles be generated from risk objectives. This is accomplished by setting the risk objectives equal to aggregate risk in the disjunctive formulation. The stressor profiles may best be generated either by setting risk-based in-stream stressor objectives (which requires a waste load allocation to generate source-specific criteria) or by iterative selection of stressor profiles and comparison of the aggregate risk to the objective. The most flexible, but computationally the most intensive solution is the iterative selection of stressor profiles.

In order to select among the infinite number of solutions, the DSMS problem is formulated as an optimisation problem that seeks to find the stressor values based on the maximum degree of acceptability of the outcome to all role players. It is proposed that regulatory satisfaction will be determined by satisfaction of the risk objective while stressor sources' satisfaction will be determined by the degree to which the stressor reduction requirement will impact on technical, economic or other issues. The overall degree of satisfaction, λ, is made up of the regulatory risk satisfaction λ_R as well as λ_{ij}, the source i, stressor j specific degree of satisfaction. The calculation of λ both as the average over all λ_{ij} and the minimum over all λ_{ij} were investigated.

The control variable was chosen as the fraction of the "raw" stressor that is allowed from the source (i.e. the stressor attenuation), x. Besides the obvious constraint that $x \in [0, 1]$ the use of an equity constraint (which requires that all stressors of the same type be treated equally), and a minimum level for λ_R may also serve as constraints. The impact of each of these has been evaluated in a hypothetical test case.

A genetic algorithm appeared to be a more effective in solving the optimisation problem than the variable simplex. The genes were composed of the set of attenuation values. The initial population of 20 individuals was selected from the randomly generated attenuation values that satisfied the constraints. The individuals were ranked according to decreased λ. The next generation was produced by sexual reproduction selecting the parents randomly from an exponential distribution and retaining the five best individuals with 15 child individuals. The new genes were generated by random crossover between parents with a mutation rate of 0.01. A published technique was used that focussed the control variable search domain after every 5 generations.

It was shown that despite the significant computation time (about 3hours for a case of 3 stressors and 4 sources on a 333MHz Pentium processor with QBASIC as coding language) satisfactory results could be obtained. From the optimal attenuation levels, source-specific stressor management criteria can be generated.

4.2 ASPECTS OF THE ECOLOGICAL RISK MANAGEMENT PROBLEM

4.2.1 BACKGROUND

Water resource management in the context of the protection of the aquatic ecosystem subject to multiple sources of a variety of stressors has to deal with:

1) the problem of setting goal-related management objectives for substantively diverse stressors and
2) an equitable and transparent apportionment of the impact among the users of the resource.

The risk assessment problem, where the risk pertains to a given combination of stressors, has to answer the question: "What is the likelihood of effect if the given stressors occur with a given likelihood?" In general the water resource risk management problem has to answer the question: "What should the stressor levels be (or stressor distribution be) if an in-stream risk target needs to be reached?" In the latter case a risk level is set and the goal and management objectives need to be derived which comply with that goal.

4.2.2 OPTIONS IN SOLVING THE DSMS PROBLEM

The diverse stressor model that had been developed in the previous chapter addressed the risk assessment question. It was shown that for ERA the conjunctive convolution of individual stressor risk could reasonably be used to estimate the aggregate risk. For stressors X all resulting in a specific unacceptable effect ($X \in \{T, Q, H\}$) the aggregate risk will be given by either version of Eq. [4.1].

$$R = \Lambda\{((E_T \mid T) \wedge T) \vee ((E_Q \mid Q) \wedge Q) \vee ((E_H \mid H) \wedge H)\}$$

With the assumption of independence this yields:

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)]$$

or: [4.1]

$$Poss(E_t \vee E_q \vee E_h) = \max\{Poss(E_t), Poss(E_q), Poss(E_h))$$

Each of these individual stressor risks is calculated from an equation of the form: $\Lambda(E_x) = \Lambda\{(\varepsilon \mid x) \wedge x\}$ where Λ is a measure of likelihood like probability or possibility. Therefore the risk is a conjunction of a SRR and a stressor occurrence.

There may now be three approaches to answer the risk management question:

a) a deconvolution of aggregate stressor risk into individual stressor risk, or
b) setting stressor-specific risk-based instream objectives., or
c) an iterative solution of the risk assessment question based on selected stressor values stopping when the aggregate risk equals the target risk (within selected precision bounds)

a) DECONVOLUTION

The deconvolution option, which seems at first appears to be the most attractive, is shown on reflection to be almost intractable. Each of the individual stressor risk terms is itself the product of two uncertain and/or variable terms, one derived from the stressor response relationship and the other from the stressor exposure. The deconvolution would therefore have to be performed in two dimensions, which decreases the tractability.

b) RISK-BASED INSTREAM OBJECTIVES

In ERA, both the SRR and the stressor can be subject to variability and uncertainty. The uncertainty in the SRR can be addressed by reducing this relationship through the assumption of a level of effect that represents in some way a minimally acceptable adverse effect. This would be analogous to using values such as the SAWQG criteria (Roux, *et al.*, 1996) except the SAWQG criteria are hazard-based rather than risk-based. In-stream stressor specific objectives, such as the South African Water Quality Guidelines (DWAF, 1996) may well reflect the regulatory goal, but is does not directly address the end-of-pipe or point-of-introduction criterion that is of importance to both the law enforcement agency and the user (discharger or abstractor). In its simplest form the quality criteria set at an in-stream point can be translated to end-of pipe values by a waste load allocation (WLA). A number of models have been used in order to accomplish this, varying from simple deterministic dilution models to stochastic dynamic models incorporating various kinetic effects (Lohani and Thanh, 1987, Chadderton and Miller, 1981, Chadderton and Kropp, 1985, Tung, 1992, Cardwell and Ellis, 1993). In principle the same may be true for water quantity or any other ecosystem stressor.

Assimilative capacity

The normal practice of waste load allocations assumes that an "assimilative capacity" exists within a receiving water body (Foran and Fink, 1993). The assimilative capacity depends on the existence of an acceptable stressor level (ASL) as a management objective corresponding to an acceptable effect level (AEL), which relates to a management goal. The capacity of the system to function "normally" in the presence of the stressor is defined as the difference between the background or natural stressor level and the ASL. This stressor "capacity" is then "allocated" among sources of the stressor.

It should be recognised that the ASL is based on assumption and its validity is therefore dependent on the validity of the assumption. Even where a natural physiological threshold exists for individual response, the natural variability within populations and between communities in ecosystems causes thresholds to uncertain quantities. Consequently, ASL is naturally uncertain and strictly only stochastic WLA methods are valid.

Problems in using generic effluent criteria

To determine what level of stressor should be allowed at the point where the stressor is induced into the system requires a set of generic effluent quality criteria (such as the "general standard" that had been applied in South Africa for a number of years (DWAF, 1986)). However, such generic stressor specific criteria, while administratively useful, do not explicitly recognise:

- The uncertainty and vagueness often inherent in ecosystem knowledge and which is dependent on expert input. Numerical management criteria are created by the projection of a set of assumptions and (possibly) value judgements onto scientific data to reduce the impact of uncertainty, creating artificial discretisations in the situation assessment space. The resulting discontinuities in situation assessments, if not used circumspectly, lead to: a) unwarranted confidence in assessment results and b) reduces the system management flexibility. Not recognising the uncertainty, variability and possibly vagueness underlying the numeric stressor-specific criteria may lead to inappropriate allocation of resources to perceived rather than real problems and induces an unnecessary conflict potential into the management process.
- The contribution of diverse stressors to the same ecological phenomenon such as loss of sustainability. This leads to the anomalous situations: a) where all stressors may comply individually and yet the management goal is not attained (e.g. Dickens and Graham, 1998), or b) the system is managed assiduously for some perceived stressors while others are not considered at all, possibly because no management criteria exist for them.
- The specific needs of users and regulators that affect the acceptability of end-of-pipe criteria. The regulatory mandate to protect the aquatic ecosystem may be perceived to be in conflict with the economic and technological constraints of the discharges. Partially, this is the result of different paradigms in which the efficacy of criteria can be assessed. Management of a river system may pit an apparently ethereal value judgement of an ecosystem against the utilitarian demands by other water users.

- Not all dischargers can achieve any given level of treatment due to economic constraints. The source- and stressor-specific upper bound to the treatment level needs to be accommodated.

c) ITERATIVE RISK ASSESSMENT

The iterative solution uses the diverse stressor risk assessment formulation iteratively with a new selection of stressor values at each iteration. It then compares the aggregate risk calculated in this way to the risk objective.

Risk in the multiple source problem

Recognising the risk principle often underlying the derivation of stressor specific criteria, a flexible management tool for deriving stressor source attenuation criteria can be created by combining ecological risk concepts with WLA. This investigation starts with the premises that:

- some stress is inevitable when water resources are being utilised,
- there may be a specific situation where stressor-specific water resource objectives are insufficient to resolve conflicting interests and the extent to which stressors need to be attenuated needs to be negotiated,
- both regulator and users are able to formulate their criteria for acceptability (for the regulator in terms of risk and for the users in terms of the degree of attenuation), and
- enough expert knowledge and/or data exist to estimate the likelihood of a common ecological end-point for all relevant stressors.

Risk objectives

Once the WLA process is in operation, the sense of effect from which it originated, is lost. The process is inclined to consider the allocation of capacity independent of effect since the allocation is done in terms of stressor metrics. Replacing the hazard-based management objectives with risk (or effect-likelihood) objectives retains the sense of *effect* management as opposed to *stressor* management. The adoption of risk objectives would help to address these issues in terms of managing multiple sources of diverse stressors.

In the context of objectives, risk:

- is used here in the sense of an expression of the likelihood of observing a specified (unacceptable) effect as a result of a stressor (such as a toxic chemical) exposure (Bartell, *et al*, 1992) and therefore explicitly recognises variability and uncertainty (Suter, 1993),
- contains elements of likelihood, target and end-point (unacceptable effect):- all of which requires explicit statement

- is able to aggregate diverse stressors (see Part 2, Paper2 and Paper 3) through its expression in terms of likelihood, and with a suitable choice of end-point, is a dimensionless expression of expectation.

The actual value of the risk objective may be a matter of policy or negotiation.

> Risk-based objectives would result in stressor specific criterion values, which are based on risk objectives, which are regulatory or societally expression of acceptability.

Discretisation of the risk continuum

The expectation of effect is assumed to have a monotonic relationship to the stressor level. This would imply that a point could be reached where the expectation is low enough to be of no further concern. This gives rise to the concept of a *de minimis* likelihood (or clearly trivial likelihood, from the legal term *de minimis non curat lex* – the law does not concern itself with trifles). Between the *de minimis* likelihood level and the *de manifestis* (or clearly unacceptable) likelihood level, there is a continuum of likelihood, which, for administrative purposes can be discretised into a series of acceptable levels of likelihood. Each of these risk objective values may itself be uncertain and only known by a clearly compliant value and a clearly non-compliant value.

4.3 FORMULATION OF THE DSMS-PROBLEM AS AN OPTIMISATION PROBLEM

4.3.1 BACKGROUND

The protection of a utilisable resource, such as water, may lead to a conflict of purpose between, on the one hand, the management agency charged with the protection of the resource and, on the other hand, the users intent on using the resource to the full. This management problem could be described in terms of a multiple objective optimisation among the conflicting goals of the role players (Sasikumar and Mujumdar, 1997). Although this is a simple problem in principle, the variability (stochasticity) and uncertainty inherent in the system and its management components are complicating factors that need at least a stochastic approach (Lohani and Thanh, 1978, Burn and McBean, 1985,Tung, 1992).

Optimisation refers to the process of finding the most favourable or best among a number of options. The solution to the diverse stressor problem proposed in the previous chapter made use of a disjunctive convolution of individual stressor risks as means of expressing the aggregate risk of the diverse stressors.

For any given value of aggregate risk, there are theoretically an infinite number of combinations of individual stressor risk levels that all result in the same aggregate risk. Each individual stressor risk level may in turn translate to an infinite number of stressor magnitude levels. If the risk-based approach to resource management is to be practical, the means need to exist to find the most favourable combination of stressor levels according to some relevant criterion.

The optimisation approach is well established in water resource management (Table 4.1)

Table 4.1 A review of optimisation techniques applied to water resource management. DO=Dissolved oxygen, BOD= Biochemical oxygen demand, COD=Chemical oxygen demand

Mathematical programming technique	Objective Function	Constraints	Special feature	Reference
Linear Programming (LP)	Cost minimisation	DO criteria	• River DO profile based in linear approximations of relevant differential equations • Mixed integer versions based on extended Streeter-Phelps model. • Parameters of the DO model, stream flow, waste flow and effluent BOD are stochastic parameters • Includes uncertainty in terms of design scenario's (see Notes)	Deininger, (1965) Loucks *et al.*, (1967) Lohani and Saleemi (1982). Hathorn and Tung (1989); Burn and Lence (1992)
Non-linear programming (NLP)	Cost minimisation Ditto	DO criteria DO criteria, seasonality of flow and treatment plant operation.	• Different river systems • Use of MINOS NLP software	Hwang, et al., (1973), Bayer, (1974). Herbay, et al., (1983).
Stochastic programming (SP)	Minimise cost	Stochastic BOD and COD	• Waste water treatment efficiency as variable	Ellis, (1987)
Dynamic programming (DP)	Minimise net cost Minimise DO deficit (Weighted objectives)	BOD constraints DO constraints	• Different waste water treatment options at each discharge point • Some use Monte Carlo simulation in water quality model	Dysart (1969), Futagami (1970), Newsome (1972), Hahn and Cembrowitz (1981), Joshi and Modak (1987).
Stochastic dynamic programming		Restrict or minimise number of standard violations Minimise magnitude of standard violation	• Use of sophisticated water quality models (WASP4 and QUAL2E) • Incorporates model (Type I) and parameter (Type II) uncertainty by regret modelling	Cardwell and Ellis (1993)
Chance constrained programming	Multi-objective: Treatment cost and water quality	Stochastic pollutant input	• Chance constraints	Boon, et al., (1989)
Fuzzy linear programming	Multi-objective (8 objectives including water quality and failure duration)	Evaluation criteria for objectives.	• Weighting of objectives • Uses fuzzy distance based ranking	Duckstein, et al., (1994)
Fuzzy chance constrained programming	Satisficing of operational risk objectives	Physical parameters of system operation	• Selection of a fuzzy risk level • Heuristic search algorithm for optimisation	Savic and Simonovic, (1991)

What is apparent in these optimisations is a) the preponderance of DO as a variable, b) the absence of ecological end-points in the problem formulation, and c) the absence of risk as a basis for optimisation.

The ecological implication of "DO deficit" is never explicitly addressed and is held as a vague and amorphous threat, which, if successfully removed, will result in some undefined benefit. The reason for the preponderance to DO modelling may be the result of two (possibly related) factors:

- The ubiquity of organic rich wastes from municipal and industrial waste-water treatment facilities, and
- The perception from legislation in many countries that oxygen depletion is the main cause of ecological stress in surface water.

While the latter may at times be a major factor determining ecosystem processes, it has also become increasingly clear that there are other stressors that are also important (See for example Dickens and Graham, (1998) and the literature cited therein).

There appears to be no alternative but to extend the optimisation process to include multiple stressors in order to solve the multiple-stressor-multiple-source problem. The optimisation problem formulation proceeds in four steps 1) formulating the philosophical point of departure, 2) isolating the pertinent stressors, 3) formulating the stressor occurrence and effect likelihood and 4) calculating the value of the objective function.

4.3.2 POINT OF DEPARTURE

It was assumed that:

1) South Africa, as an semi-arid, relatively poor country with a dependence on ecotourism would require that water resources be managed for maximum return flow, minimum stressor attenuation while striving to attain ecological protection goals. All of these requirements are of course not generally true, but it represents a precautionary scenario.
2) There exists enough goodwill and a spirit of co-operation between regulator and regulatees to solve the catchment management problem and for both parties to be willing to objectively formulate acceptability criteria in order to reach a compromise solution and that, above all, the regulatory framework allows for such a compromise.
3) The solution to the problem will be determined by the goal directed considerations informed by technology and economic considerations.

The implications of this point of departure is that:

a) All wastewater needs to be returned to a surface water resource. The National Water Act demands that no user may impair the sustainability of the water resource and, therefore, the contaminants in water that impact on the aquatic ecosystem need to be attenuated,

b) The best available technology from a Developed World point of view may not always be available to each stressor source and that homogeneous stressor attenuation levels may not always be feasible although it would the ideal,

c) Socio-economic or other "soft" (non-technical) factors may influence the extent and level of stressor attenuation and water resource protection (Beck, 1997). Each level of stressor attenuation carries with it an implication for the users and the ecosystem. These implications are likely to be interpreted in terms of diverse and possibly incompatible metrics. For example, the discharger may interpret a reduction of the allowable discharge of toxic substances in terms of treatment cost, employment opportunities lost as a result of inability to meet regulatory standards etc. On the other hand, the regulatory authority, charged with the protection of the aquatic ecosystem, interprets the attenuation level in terms of the threat to the long-term sustainability to the system. If the metrics of interpretation are not brought onto a common footing, the conflict may become irresolvable.

d) One source of communality between the user and the regulator is the acceptability of the regulated situation. The acceptability of different levels of stressor attenuation is likely to be epistemic so that it can best be described by a fuzzy set. This implies that acceptability can be graded in terms of degree of acceptability or conformity to the descriptor "acceptable".

e) The style of management on the part of the regulator would allow for explicit goal-oriented management and that these goals can be captured in risk values.

4.3.3 OBJECTIVE FUNCTION AND CONSTRAINTS

Generically, optimisation requires two components: an objective function expressing which values are to be minimised or maximised and (optionally) the constraints under which the optimisation should operate. The format of the problem would be:

> Maximise (minimise) the OBJECTIVE, which is a FUNCTION of CONTROL VARIABLE
>
> So that CONSTRAINTS are satisfied

OBJECTIVE FUNCTION

For the formulation of an objective function, communality between the regulator and the regulatee needs to be established.

- Under the NWA, the regulator is primarily concerned with the protection of aquatic ecosystem and this could be expressed in terms of the minimisation of ecological risk.

- The regulatee would have socio-economic and technical considerations as prime concern.

> The extent to which each role player is satisfied with the outcome of the regulatory process, is a common denominator in the sense of representing a common measure. This degree of satisfaction is designated by λ; the degree of satisfaction obtained with the level of risk achieved λ_R, while the degree of satisfaction of the manager of source *i* with the regard to stressor *j* is λ_{ij}.

The satisfaction of all regulatees can be aggregated into λ_x. The value of λ_x could be derived in two different ways:

> Option 1: The minimum acceptability over all controllable stressors at each source could be calculated and the average could be calculated over all the sources in the reach
>
> $$\lambda_x = \frac{\sum_{k=1}^{n} \min_k\{\lambda_{,i,j}\}}{n}$$ for *n* control variables, or
>
> Option 2: The individual attenuation acceptability could be aggregated conjunctively, in which case:
>
> $\lambda_x = \inf\{\min\{\lambda_{ij}\}\}$ for each stressor *i* and source *j*.

CONTROL VARIABLES.

The control variable need to express those entities that can be changed by the manager/ decision-maker in order to achieve the goal set in the objective function. There are two possible common denominators suggested by the objective function: the stressor levels and the degree of attenuation of the "raw" stressor levels. The advantage of the degree of attenuation is that it is unitless.

> The choice of control variable is the degree of attenuation of the "raw" stressor, designated by *x*. Each stressor *i* and source *j* combination is given a value x_{ij}.

CONSTRAINTS

The constraints describe the limits within which the optimisation must be performed. These might include physical constraints and process constraints. The might include the physical limitations on the value of the objective function, control variables or any other parameters involved in them. It may be required in the interest of being fair and equitable, that all similar stressors should be treated similarly.

> The generic constraints chosen for this study are:
>
> The attenuation levels by definition are defined such that $x_{ij} \in [0, 1]$ or $0 \leq x_{ij} \leq 1$.
>
> The degree of satisfaction is defined such that $0 \leq \lambda \leq 1$.
>
> It may be required that a maximum risk ρ is specified which may not be exceeded, therefore $\lambda_R \leq \rho$ and $0 \leq \rho \leq 1$
>
> Optionally an equity constraint may be formulated such that for a stressor i from sources k and l the absolute difference between the attenuation of s from these source must always be less than an amount δ, i.e. $|\,|x_{ik} - x_{il}|\,| \leq \delta$. δ is defined in Eq.[4.8]

Fuzzy constraints

- In order to produce such a general acceptability criterion, the user that may incorporate his own particular weighting of cost and technological implications of a treatment level x_{ij}. This requires at least an expression from each resource user of an acceptability pair $\{x_{ij}^{min}, x_{ij}^{max}\}$. Here, x_{ij}^{min} represents a treatment level that is completely acceptable, while x_{ij}^{max} represents a treatment level which, for whatever reason, is completely unacceptable.

- For this study it has been assumed that between these two levels (and possibly even including these levels) there exists a continuum of acceptability. Without loss of generality a stepped function could also have been used as long as the function is monotonic.

- Likewise, the regulator defines a fuzzy risk acceptability criterion by specifying (possibly resource dependent) *de minimis* and *de manifestis* risk levels, ρ^{min} and ρ^{max} respectively.

CALCULATION OF RISK/CONCERN VALUES

The ecological risk or concern, ρ, is calculated from the likelihood of the stressor occurrence and the cumulative likelihood of effect on exposure to a stressor. This requires either (1 OR 2) AND 3:

1. Measurement of the stressor values in-stream over a suitable spatial and temporal domain and estimating the likelihood of stressor occurrence from stressor observation data,
2. Modelling the stressor occurrence likelihood,

3. Estimating the stressor response likelihood from laboratory or field data.

Estimating stressor occurrence likelihood

Generally, the in-stream stressor value s_i will be a function of the unattenuated stressor value, s^0_{ij}, the treatment level, x_{ij}, the apparent stressor specific degradation constant, k_j, and the retention time τ_i between stressor entry point and the point of interest (see Appendix).

The ideal would be to estimate stressor occurrence likelihood from measured data. This is unlikely in the case of *ab initio* calculation of stressor attenuation. It is more likely that the second requirement can be met. Models of different levels of sophistication and environmental realism exist to calculate in-stream water quality parameters (e.g. CEAM, 1996). Predictive hydrological models also exist that estimate the in-stream flow from rainfall data (e.g. Pitman, 1973). Of the stressors selected for this study only the habitat degradation remains to be assessed *in situ*, but methods do exist to perform such an assessment (e.g. Kleynhans, 1996b).

For a probabilistic risk assessment, it is important that a stressor occurrence model be able to simulate the impact of temporal/spatial variability as well as model and/or parameter uncertainty. A common method to this is by Monte Carlo simulation. Possibilistic models would need to be able to deal with fuzzy inputs.

Two problems were encountered with the models that could be used for toxic substance models: 1) The software code for the models was not readily available, and 2) Few of the available models have the ability to accept or generate stochastic data. It was therefore difficult to integrate these models with rest of the coding used here. For the purpose of this study, a simple dilution model with constant first-order degradation kinetics was used to calculate the concentration of toxic substances, while it was assumed that the flow distribution was known *a priori*. A possibilistic model is described in 4.4.2. A stochastic analogue using Monte Carlo simulation was also attempted (coding appears in the Appendix of this chapter). This model was not pursued further for two reasons: the nature of the ecological impact favoured an epistemic approach to stressor occurrence that necessitated a possibilistic rather than a probabilistic methodology and the coding language used could not easily resolve the computer memory management problems encountered in the Monte Carlo simulation.

In most cases the stressor possibility distribution will be identical to the stressor variable distribution for example, in the case of toxic substances, the toxic stressor distribution will be identical to toxic substance concentration expressed as toxic units. However, in the case of flow, the flow itself is not the stressor, but flow insufficiency is more likely to be. In this case, the stressor possibility distribution derives from the extent to which the flow possibility distribution, $\mu_Q(q)$ can be said to be descriptive of the state of flow insufficiency, $\mu_I(q)$, and therefore: $\mu_S(q) = max\{min(\mu_Q(q), \mu_I(q)\}$. Here, the flow insufficiency is estimated from q^{min}, a level of flow below which organisms would likely succumb completely to the end-point effect and q^{max}, a level above which no end-point effect would be observable (see Appendix).

4.3.4 FORMULATIONS OF DSMS-OPTIMISATION PROBLEM

The conflicting needs of role players in a catchment was addressed by Tung (1992) in using multiple-objective WLA (involving the optimisation of conflicting needs to constraints) as an example of the application of multiple objective optimisation problems (MOOP's). Here the single objective concept of optimality is no longer valid. Unless a prior knowledge exists to weigh the conflicting objectives, the solution to the MOOP remains a locus of points representing a trade-off. The concept of optimality is replaced by the 'non-inferior solution' which is corresponds to a curve or surface until the decision-maker supplies the weighting. Chang *et al.*, (1997) applied fuzzy interval multiobjective optimisation to water pollution control in a river catchment showing that different types of uncertainty can be combined through a possibilistic approach. In general, these only consider water quality management in terms of discharge objectives.

In practice, the optimisation then involves finding the stressor and source specific treatment levels that maximises the acceptability parameter λ (or alternatively minimises the unacceptability $(1-\lambda)$)

CRISP FORMULATIONS

The optimisation problem may be formulated in several ways involving issues that may be of concern to the stakeholders, such as protection of the ecosystem, stressor reduction cost, and treatment equity among different stressor sources. From an ecosystem protection point of view the optimisation problem might be formulated as:

1. Minimise the cost of ecological concern (or risk) reduction by setting the stressor reduction level x_{ij} for the i^{th} stressor from the j^{th} source to a value that will satisfy an upper ecological risk limit for the system as well as possible technological or other ethical constraints.

2. Minimise the ecological concern (or risk) to the system by adjusting x_{ij} so as to meet cost, technological and ethical constraints.

3. Zimmermann's approach: maximise the degree of satisfaction of all stakeholder goals within given cost and risk constraints (Lai and Hwang, 1994).

$$
\begin{aligned}
&\text{maximise } \lambda \\
&\text{so that } \mu_R(x_{ij}) \geq \lambda && \text{(Regulatory goal)} \\
&\qquad \mu_G(x_{ij}) \geq \lambda && \text{(User goal)} \qquad [4.2] \\
&\qquad x_{ij} < \gamma_{ij} && \text{(Technological constraint)} \\
&\qquad ||\, x_{ik} - x_{il} \,|| \leq \delta && \text{(Equity constraint)}
\end{aligned}
$$

(Where γ_{ij} is the technological constraint for stressor *i* at source *j* and δ is a maximum allowable difference in attenuation level for stressor *i* between any two sources *k* and *l*).

In the first formulation it is assumed that it is feasible to estimate the financial cost as a function of x_{ij} quantitatively (Burn and McBean, 1985). Given that the unattenuated stressor magnitudes may in general be uncertain or variable, it would be necessary to set a compliance level α (say α = 0.95) and calculate the corresponding x_{ij}. The difference between the first two formulations of the problem is the aspect on which compromise has to be made. From a purely utilitarian point of view the second formulation is preferred while from a purely protective point of view the first formulation is preferred. However, both formulations require a functional relationship between constraints and control variables, but this is often lacking (Lai and Hwang, 1994).

FUZZY FORMULATION

A fuzzy set equivalent of this optimisation problem (Eq. [4.2]) could use the Bellman-Zadeh fuzzy decision (Z) which is defined as the intersection between fuzzy goals (G) and fuzzy constraints (C) (DuBois and Prade, 1994, Klir and Yuan, 1995), i.e. $Z = G \cap C$. This represents those goal and constraint values that satisfy both sets. The distinction between the goals and constraints is lost.

- The objective function supposes that each stakeholder will compromise on its constraint requirements and will be able to express its satisfaction with the consequence of a value of x_{ij} in terms of a satisfaction parameter λ.
- For resource protection, the protection agency may impose a risk level ρ_0, but will compromise that to the extent ρ'.
- Each stressor source may wish to reduce their expenditure for stressor reduction to a minimum. Each stressor source may set an ideal limit c_j but will compromise to the extent c'_j.

This translates the fuzzy programming formulation (Eq. [4.2]) to a crisp programming formulation (Eq. [4.3].

$$
\begin{aligned}
&\text{Maximise} \quad \lambda \\
&\qquad \text{So that } C_j(x_{ij}) \leq c_j + c'_j(1 - \lambda) \\
&\qquad\qquad R(x_{ij}) \leq \rho_0 + \rho'(1+\lambda) \qquad [4.3]
\end{aligned}
$$

$$x_{ij} < \gamma_{ij}$$

$$||\, x_{ik} - x_{il} \,|| \leq \delta$$

An interactive inexact fuzzy multiobjective programming (IFMOP), which is more extensive version of Eq. [4.3], was used (Wu *et al.*, 1997) in the water pollution control planning of a lake where the economic activities in the catchment had been specifically included. A problem that arose in this case related to separating objectives that had to be maximised from those that had to be minimised. In this case this difficulty does not arise since there is only one objective that needs to be maximised.

Application of the fuzzy formulation approach along with the constraints and terminology of 4.3.3 to Eq. [4.3] produces the model Eq. [4.4]:

Minimise (1-λ)

So that $$\lambda = \begin{cases} 0 & if\, \lambda_R < \zeta \\ \min\{\lambda_R, \lambda_x, \lambda_{eq}\} & if\;\; \lambda_R \geq \zeta \end{cases} \qquad [4.4]$$

$$x \geq 0$$

and λ_R, λ_x and λ_{eq} as defined below in Eqs. [4.5], [4.6] and [4.7]. The parameter $\zeta \in [0, 1]$ is a minimum risk compliance level required by the regulator. The ecological risk with reference to the chosen level of organisation and end-point, ρ, is calculated from the possibility distribution of the stressor ($\mu_S(s_i)$) and the possibility distribution of the effect over the stressor range ($\mu_E(s_i)$).

The satisfaction terms in the optimisation model were calculated as follows:

$$\lambda_{ri,j} = \begin{cases} 1 & if\; \rho_{i,j} < \rho_{i,j}^{\min} \\ \dfrac{\rho_{i,j}^{\max} - \rho_{i,j}}{\rho_{i,j}^{\max} - \rho_{i,j}^{\min}} & if\; \rho_{i,j}^{\min} \leq \rho_{i,j} \leq \rho_{i,j}^{\max} \\ 0 & if\; \rho_{i,j} > \rho_{i,j}^{\max} \end{cases} \qquad [4.5]$$

$$\lambda_{ij} = \begin{cases} 1 & if\; x_{i,j} < x_{i,j}^{\min} \\ \dfrac{x_{ij}^{\max} - x_{ij}}{x_{ij}^{\max} - x_{ij}^{\min}} & if\; x_{ij}^{\min} \leq x_{ij} \leq x_{ij}^{\max} \\ 0 & if\; x_{ij} > x_{ij}^{\max} \end{cases} \qquad [4.6]$$

$$\lambda_x = \min\{\lambda_{ij}\}$$

$$\lambda_{eq} = \begin{cases} 1 & if \delta < \varepsilon_{\min} \\ \dfrac{\varepsilon_{\max} - \delta}{\varepsilon_{\max} - \varepsilon_{\min}} & if \varepsilon_{\min} \le \delta \le \varepsilon_{\max} \\ 0 & if \delta > \varepsilon_{\max} \end{cases} \qquad [4.7]$$

where $\rho = \max\{\min(\mu_S(s_i), \mu_E(s_i))\}$, $s_i = f(s_{ij}^0, x_{ij}, k_i, \tau_j)$ and

$$\delta = \max_i^m \left(\frac{\max_i^n\{x_{ij}\} - \min_i^n\{x_{ij}\}}{(\max_i^n\{x_{ij}\} + \min_i^n\{x_{ij}\})/2} \right) \qquad [4.8]$$

4.3.5 SOLVING THE OPTIMISATION PROBLEM

A large number of optimisation algorithms are available, of which two were selected as being conceptually simple as well as relatively easy to encode so that it could be effectively combined with a suitable objective function evaluation. The two that were eventually selected are the variable simplex and genetic optimisation algorithms.

THE VARIABLE SIMPLEX ALGORITHM

The Simplex algorithm (Nelder and Mead, 1965; Lowe, 1967; Betteridge, *et al.*, 1985; Gill *et al.*, 1991) is a heuristic search algorithm based on the projection of a simplex, which is a (n+1)-dimensional geometric figure for an n-dimensional search space. The objective function is evaluated at each of the n+1 vertices of the figure and a new figure is generated by projecting the worst vertex through the centre of gravity of the remaining n vertices. The Variable Simplex algorithm (Fig. 4.1) allows for contracting or expanding the projection in the Simplex algorithm to achieve a more rapid convergence to the optimum. Since this algorithm may be stuck at a local optimum, it is suggested that the search be restated at a different set of starting values. The algorithm as described by Shoup and Mistree (1987) was used.

GENETIC ALGORITHMS

Genetic algorithms (GA's) belong to the family of random search algorithms with a focussing heuristic (Bäck, 1996). GA's have as their basis the principles of Darwinian evolution. The mechanisms of GA's are similar to those in population genetics and are based on exchange of genetic material between individuals to produce new individuals whose suitability may differ from those of the parent individuals. The main operations are selection, exchange, mutation and reproduction. It is also possible to impose search heuristics to speed up the convergence. The version used here is of the elitist type where the best performing individuals are selected along with the offspring to compete in a tournament to find the best performing individuals.

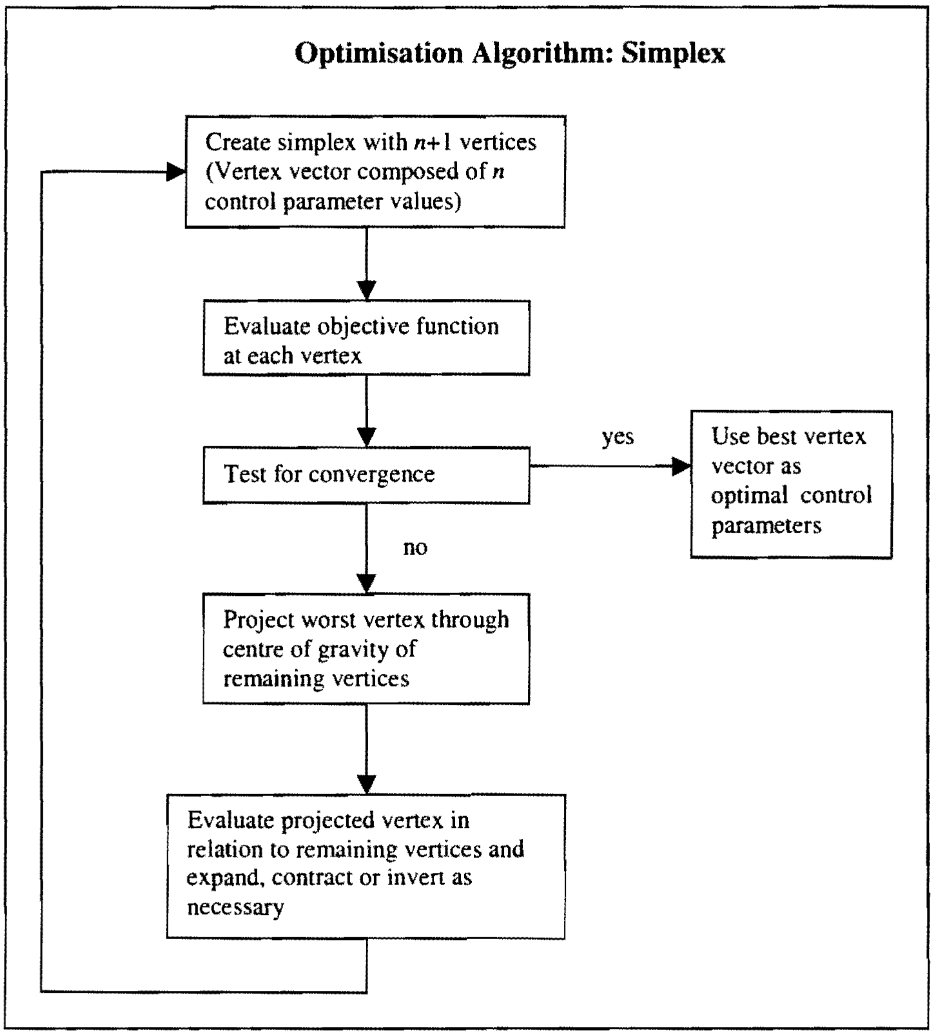


Figure 4.1. Diagram of the variable simplex algorithm.

4.4 HYPOTHETICAL CASE STUDY

The use of optimisation as a means to generate risk-based source criteria is investigated using a hypothetical test case. The parameters used in this case were not taken from any specific study, but represent considerations from a number of sources typical of situation in which such a method might be used.

The optimisation algorithms are first evaluated against a test (Colville response surface as described in Shoup and Mistree (1987)) where the optimum is known (Scenario 1). The genetic algorithm was then used to evaluate source specific criteria in three different scenarios resulting in different objective functions. In each of the last three scenarios two options for initialising the algorithm is evaluated. Some of the results are listed in the Appendix.

4.4.1 SELECTING STRESSORS AND SRR'S

The stressors chosen for the hypothetical case study are:

TOXIC SUBSTANCES

Although no specific general data on the occurrence of toxic substances in fresh water in South Africa were available, some problem related studies indicated that toxics do occur periodically in surface water. Chlorination is still a common practice on treated sewage effluent before discharge to surface water in South Africa (Williams, 1996). Toxicity assessments on chlorinated sewage from treatment plants in the KwaZulu-Natal province of South Africa indicated that it may an important contribution to surface water toxicity (Williams, 1996). The instream concentration of toxic substances will generally be a function of the input load to total load ratio, and will therefore be dependent on flow. It was further assumed that toxic concentration would be determined for point sources by a suite of whole effluent toxicity (WET) assessments. From the toxicity assessment data a concentration suitable to the end-point for the management goal will be selected e.g. a no-observed-effect concentration (NOEC) at a discrimination level $\alpha 1$. The level of the toxic stressor in the effluent, x, is expressed as toxicity units (TU's), which is calculated as: x (in TU's) = (the actual concentration of the effluent)/(NOEC) (Suter, 1993). The response curve for the risk assessment is simulated from the response curve from which the NOEC was calculated such the expected response y would be given by:

$$y = \frac{1}{1 + A_0 e^{-bx}} \qquad [4.9]$$

The constants A_0 and b are determined by solving [4.1] with the conditions that if $y = \alpha 1$ then TU = NOEC and if $y = \alpha 2$ then TU = $b2$ where $b2 = b$/NOEC and b is the concentration corresponding $\alpha 2$ in the original curve.

HABITAT DEGRADATION

Although no generic data were available for the South African status of instream habitat degradation as a stressor, some results (Sparks and Spink, 1998; Kleynhans, 1999b) seem to indicate that on a site-specific basis this may a major stressor to the aquatic ecosystem. Habitat degradation as a stressor must be distinguished from flow related habitat insufficiency, which was considered to be related to flow insufficiency (Milhous, 1998). As used here, habitat degradation refers to physical removal of aquatic habitat components, so that even when flow as represented by water depth or flow rate is sufficient, there is simply inadequate habitat to support aquatic life. No specific data on habitat stress assessment was found although the importance of habitat is recognised (Hardy, 1998; Lamouroux, *et al.*, 1998; Kleynhans, 1999a). The assessment of the response of aquatic organisms to physical habitat degradation has to be performed by a competent aquatic ecologist. The response curve may be estimated from a no-observable effect level of habitat degradation and an unacceptable level of habitat degradation corresponding to a threshold level below which no effect is expected and a level above which effects are certain to occur. The response may be simulated by a trapezoidal function or an s-shaped response from a function similar to Eq. [4.9].

FLOW INSUFFICIENCY

Water as the major habitat of aquatic organisms, needs to be maintained at a seasonally appropriate level for the aquatic ecosystem to remain functioning healthily (King and Louw1998; Moyle, *et al.*, 1998; Kleynhans, 1999b). In many cases the water depth is important as it provides access to specific habitat such as pools or riffles, which are important in the life histories of specific organisms. In some cases, the flow rate is important (Sparks and Spink, 1998). Flow insufficiency as a stressor does not include naturally occurring floods or droughts. Aquatic organisms in semi-arid countries may well have adapted to such events (Davies, *et al.*, 1994). Flow stress has, for the sake of illustration, been designated as (expected flow – actual flow)/(expected flow).

4.4.2 PROBLEM STATEMENT

Consider a river reach with three discharges and one abstraction. The magnitude of stream flow is representative of a small stream that already has significant toxicity present upstream of the reach being modelled. The discharges to this stream are typical of small sewage treatment works (about 1 megaliter per day). The toxicity, expressed as toxicity units, is based on chronic toxicity values and is not unlike those obtained for a small impacted stream in an industrialised area in South Africa (Jooste and Thirion, 1999). The habitat stress is assumed to derive mostly from streambed modification through farming and construction activities. Although streams of this magnitude are not significant as major water suppliers, they are typical of those that may be the refugiae and possible sources of recolonisation for larger streams and rivers and may be worthy of being protected for this reason.

The stream is modelled as a system with four nodes (see Figure 4.1) with inputs and outputs. The first two nodes receive discharges, the third node yields abstraction and the fourth node receives discharge. The habitat stress is associated with the node upstream of the stressed habitat.

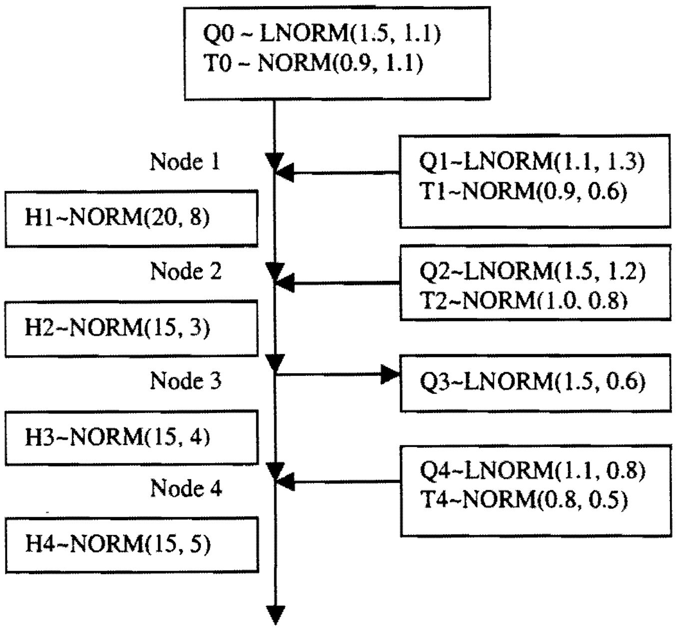


Figure 4.1 A diagram illustrating the set-up of the hypothetical test case. The input values for the stochastic problem formulation are shown. For a median x and standard deviation y, LOGNORM(x,y) indicates the lognormal distribution and NORM(x,y) indicates a normal distribution.

The control variables are:

1) the attenuation of the volume of water abstraction (xQ3),
2) the attenuation of toxic substances at the discharge nodes (xT1, xT2 and xT4) and
3) the habitat stress attenuation at each node (xH1, xH2, xH3, xH4).

The discharge flows, the discharge toxic concentrations, the habitat stressor levels as well as the upstream flow and toxicity levels are considered stochastic variables. It is assumed that the toxicity in the river is subject to degradation following a simple exponential decay function. The toxic levels at each node are calculated by mass balance (Eq. [4.10]).

$$sd_{i,j} = \frac{su_{i,j} \cdot qu_j + (-1)^{z_j} \cdot q_j \cdot (1 - f_{1,j})^{z_j} \cdot \{s_{i,j-1} \cdot z_j + s_{i,j} \cdot (1 - f_{i,j}) \cdot (z_j - 1)^{z_j}\}}{qd_j}$$

$$qu_j = qd_{j-1}$$

$$qd_j = qu_j + (-1)^{z_j} \cdot q_j \cdot (1 - f_{1,j})^{z_j}$$

$$su_{i,j} = sd_{i,j-1} \cdot \exp(-k_{i,j-1} \cdot \tau_{j-1})$$

[4.10]

where $i \in$ {dilution dependent stressors}, $j \in$ {sources}, f_{ij} is the attenuation factor and $z_j = 1$ for an abstraction and 0 otherwise. In the hypothetical case $i \in$ {T} and $j \in$ {1, 2, 3, 4} and $z_3 = 1$. For control variables $f_{ij} \in [0, 1)$ (i.e. $f_{ij} = x_{ij}$), while for non-control variables $f_{ij} = 0$.

The in-stream habitat degradation values remain unaltered over time but can be attenuated.

Table 4.1. Numerical input values for the model described in the text (Scenario 2).

Parameter	Upstream	Point 1	Point 2	Point 3	Point 4
Flow median q^0	4	1.1	2.1	2.5	1.8
Flow std dev.	1.1	1.15	1.28	1.56	1.11
Tox units median t^0	0.3	0.8	1.1	0	0.9
Tox units std dev	0.1	0.21	0.34	0	0.26
Habitat degr. Min	0	10	15	10	20
Habitat degr. Med	10	20	30	30	30
Habitat degr. Max	20	30	40	50	50
Q^{min}	-	1.5	1.5	2	2
Q^{max}	-	2.1	2.1	2.5	2.5
Flow stress effect min	-	2.2	2.5	2.5	2.6
Flow stress effect max	-	3.5	3.5	3.5	4
Tox stress effect min	-	0.2	0.2	0.2	0.2
Tox stress effect max	-	1.1	1.1	1.1	1.2
Habitat stress effect min	-	30	30	30	30
Habitat stress effect max	-	75	75	65	75
Tox degradation constant k (day^{-1})	-	0.2	0.2	0.2	0.2
Retention time τ (days)	-	2	3	2.5	4
Treatment acceptability					
x_q^{min}	-	-	-	0	-
x_q^{max}	-	-	-	0.6	-
x_t^{min}	-	0.2	0.2	-	0.3
x_t^{max}	-	0.7	0.8	-	0.75
x_h^{min}	-	0	0	0	0
x_h^{max}	-	0.15	0.1	0.1	0.2
Regulator risk acceptability					
ρ^{min}	-	0.05	0.05	0.05	0.05
ρ^{max}	-	0.15	0.15	0.15	0.15

4.4.3 METHODOLOGY

GENETIC ALGORITHM

In order to create the genetic material, the initial values for the control parameters in the optimisation problem were encoded as 16-bit binary numbers. All values were multiplied by 1000 and truncated to integers. The gene characterising an individual was created by the concatenation of the 16-bit binary numbers.

The genetic algorithm is outlined in Fig. 2. In the genetic algorithm, the vector of control parameters was considered as a part of a "chromosome" characterising an "individual" solution to the optimisation problem. The control parameter values were multiplied by 1000 to give a value in the interval [0, 1000]. This was done in order to facilitate the conversion of control variables to binary format.

From these two parents an initial population of 16 individuals (including the parents) were generated each with its own chromosomal values, by methods as described in the Appendix. These were then converted to binary numbers and encoded into a 16-bit string for each of the control parameters. The genetic algorithm used in this study was of the "elitist" type where the four best parents were preserved as part of the next generation. The parents were selected randomly with an exponential probability distribution (location parameter = 1).

The crossover was selected so that each 16-bit byte had an equal chance of being selected from either parent. Mutations, where the 0's and 1's were inverted on transcription of the parent bit to the child bit, were performed with a probability of 0.1.

The performance of the each individual in the population was determined by decoding the chromosome into control parameters and recalculating λ. The population was then rearranged from best to worst, based on the λ values.

After every epoch of 40 generations the control parameters were re-initialised from a suitable distribution and this process was repeated for 10 epochs. This cycle was repeated 10 times.

The performance of the best individual in the population was recorded, as were the values of the control parameters corresponding to the best performing individual in the population. In order to speed up the process both the range of the search domain and a heuristic adaptation the direction of search for each control variable was performed after every 5 generations (Ndiritu and Daniell, 1999). After refocusing and adaptation the population was reinitialised.

Methods used for the assignment of control variable values in the genetic algorithm:

(a) For initialisation, two parent individuals are generated by random assignment of control variable values from the interval [0, 1] by different distributions. The individuals are selected on the basis of producing a value $(1-\lambda) < 1$. The control variable values for the initial population are generated from the parent values by the random addition of $\pm(0.3*$ the parent value) to the parent value.

(b) For the re-initialisation of control variable values after each epoch or after refocusing, the tournament population was generated by assigning the values from the variable specific interval $[x_i^{min}, x_i^{max}]$ by exponential distribution with location parameter μ where $\mu = 2\ln(0.5)/(x_i^{max} - x_i^{min})$.

The two options in assigning the control variable values in the initialising and re-initialising steps are:

- Option 1: initialise from a uniform distribution and re-initialise from an exponential distribution and
- Option 2: initialise and re-initialise from an exponential distribution.

TESTING OF ALGORITHMS

The performance of both algorithms were tested by obtaining the minimum of the four parameter Colville response surface described by Shoup and Mistree (1987).

The fundamentals of the methods for the Variable Simplex and GA used are described in Shoup and Mistree (1992) and Ndiritu and Daniell (1999) respectively. The coding of the methods was tested by using the Colville response surface and establishing whether the optimum point could be reached.

Table 4.2 Parameters for the evaluation of coding for the simplex and genetic optimisation algorithms.

Parameter	**Value**
Simplex: Expansion coefficient α	1.0
Contraction coefficient β	0.5
Contraction coefficient γ	0.5
Genetic algorithm	
Number of cycles (s)	10
Number epochs per cycle (e)	10
Number of generations per epoch (g)	40
Number generations for focussing (g1)	5
Number of generation for heuristic shift (g2)	5
Probability of mutation (m)	0.1

The hypothetical test case was then coded in Microsoft® QBASIC and run on a 333 MHz PentiumII processor with parameters as set out in Table A4.2 in the Appendix to Chapter 4. For the genetic algorithm, the basic algorithm and attempted improvements as well as the respective coding appear in the Appendix.

Both simplex and genetic algorithms found the theoretical extremum within about 50 iterations. However, application of the simplex algorithm failed to converge in the hypothetical case above.

CALCULATING STRESSOR VALUES

The procedure followed in the calculation of point source stressor attenuation values is outlined in Fig. 4.2. The characteristics of the three sources of discharge and one abstraction are shown in Table 4.1 (Scenario 2). The calculations were repeated with two other scenario's where the acceptability range for Source 1 was changed to $x \in (0, 0.3]$ (Scenario 3) and another where the risk acceptability was changed to $\rho \in [0.01, 0.05]$ (Scenario 4).

Generating possibility distributions

Instead of treating the inputs to the mass balance equation (used to calculate the toxicity levels from stochastic inputs) as a stochastic quantity, it was interpreted as a deterministic variable that is subject to epistemic uncertainty. For the purpose of this calculation the probability distributions were treated as possibility distributions by normalising to the maximum of the probability distribution (i.e. the possibility that $X = x$, $\Pi(X=x) = P(X=x)/P(X=\text{mode } x)$).

The calculation of the fuzzy toxicity level was then performed by considering nested sets of intervals based on α-cuts of the stressor possibility distributions (Kaufman and Gupta, 1985; Klir and Folger, 1988), using interval arithmetic (Alefeld and Herzberger, 1974). The possibility range of each variable was discretised into 20 values (including 0 and 1). The upper and lower bound toxicity levels were calculated at each α-level, which corresponds to an upper and lower risk level. The risk satisfaction level ,λ_R, was calculated from the maximum risk and the risk acceptability values ρ_{min} and ρ_{max}. In order to counter the possible degeneracy induced by the fuzzy objectives in Eqs.[4.5] and [4.6], values ρ^{min} and ρ^{max} and x_{ij}^{min} and x_{ij}^{max} were used as the abscissa values corresponding to the ordinate values of 0.05 and 0.95 respectively in Eq. [4.9], while q^{min} and q^{max} were used as the abscissa values corresponding to the ordinate values 0.95 and 0.05 respectively in Eq. [4.11].

$$y = \frac{A_0 e^{-bx}}{1 + A_0 e^{-bx}} \qquad [4.11]$$

The control parameters were selected as those attenuation values that were actually controllable. The abstraction concentration and the effluent flow attenuation were not considered to be practically controllable. This resulted in eight control parameters being used, i.e. $x_k \in [0, 1]$, $x_k \in \{f_{ij}\}$, $i \in \{Q, T, H\}$ and $j \in \{1, 2, 3, 4\}$ for the test case.

ESTIMATING THE INFLUENCE FUZZIFICATION PARAMETERS

To estimate the effect a change in acceptability parameters will have the toxic attenuation acceptability parameter for source 1 and risk acceptability parameters were adjusted as shown in Table 4.3.

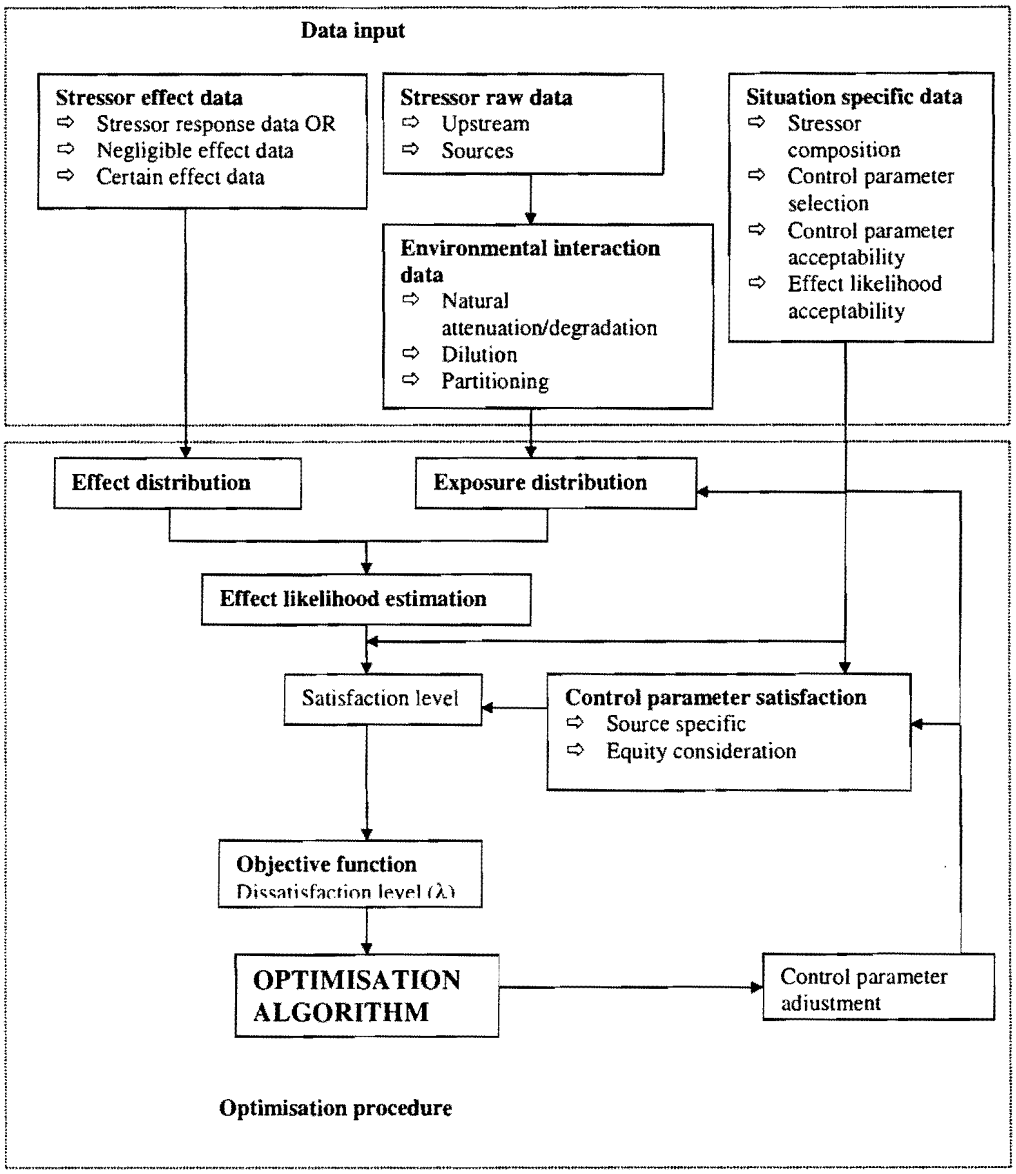


Figure 4.2. An outline of the methodology used to calculate the stressor attenuation levels.

Table 4.3. Acceptability parameter values for Scenarios 2, 3 and 4

Scenario	$x_{t,1}^{min}, x_{t,1}^{max}$	ρ^{min}, ρ^{max}
2	0.2, 0.7	0.05, 0.15
3	0.01, 0.3	0.05, 0.15
4	0.2, 0.7	0.01, 0.05

Optimisation Algorithm: Genetic algorithm

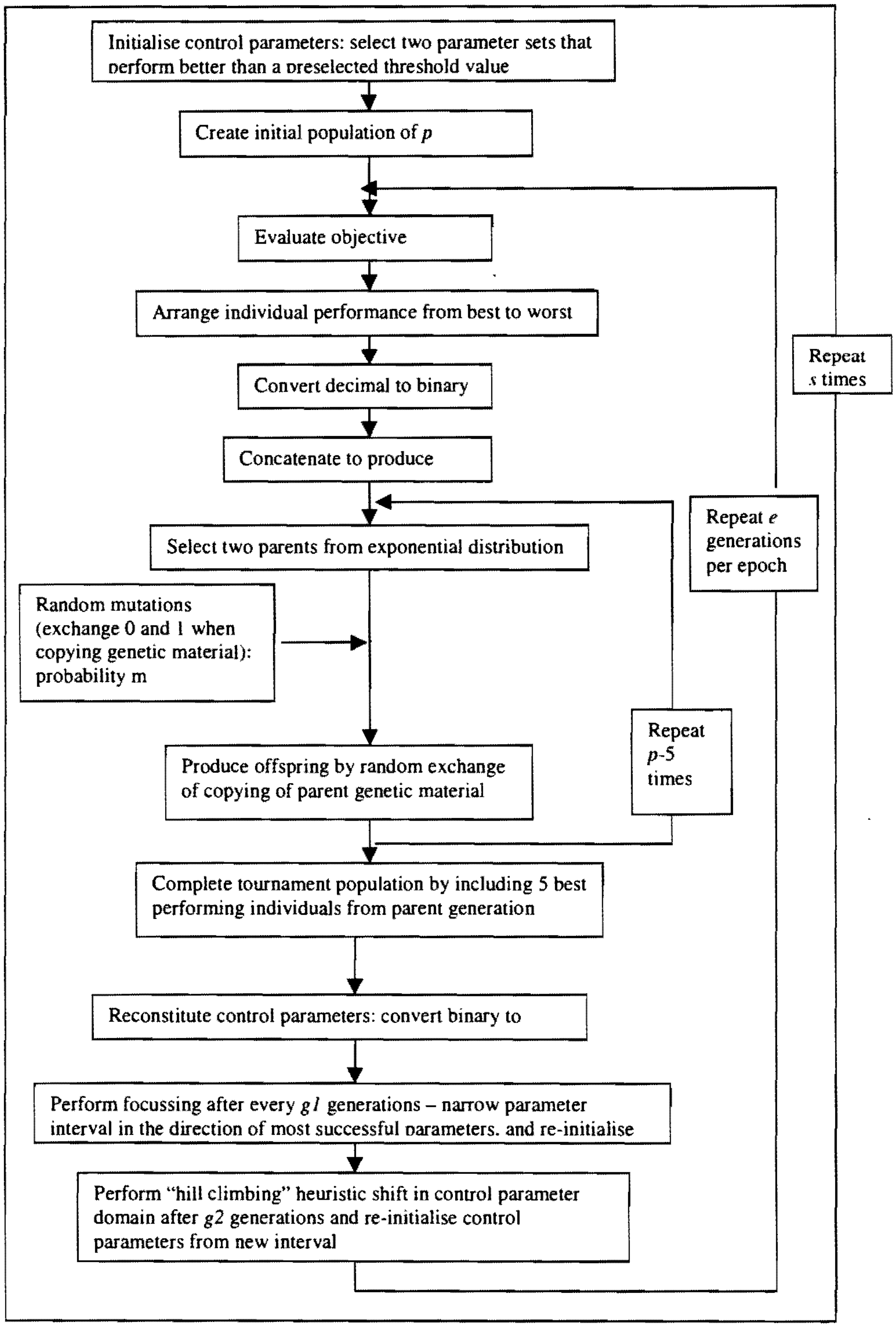


Figure 4.3. An outline of the genetic optimisation algorithm used in the estimate the attenuation levels for multiple stressors.

4.4.4 RESULTS

COMPARISON OF ALGORITHMS

The results for the comparison between the Variable Simplex and GA optimisation appears in Table 4.4.

Table 4.4 Results of the coding tests on the Colville response surface.

Algorithm	Result
Variable Simplex	Convergence dependent on choice of initial values. With favourable choice of initialising values converges in 40 to 50 iterations to within 10 in 1000 000, i.e. about 200 to 400 evaluations of objective function. One hundred repetitions of the process with random initial values did not produce one case of convergence.
Genetic Basic	Convergence independent of initial values if total number of generations > 100 and initial population ≥ 4*number dimensions, i.e. > 2 000 evaluations of objective function. Ten repetitions of the process produced six cases of convergence. (Parameter values found by trial and error.)

The result for the Variable Simplex algorithm is different from that obtained by Shoup and Mistree (1987) who obtained convergence for the Colville response surface irrespective of the initialising values of the control parameters. The reason for this difference is not immediately apparent. It was assumed that some coding error must have caused this difference, but meticulous checking of the coding did not reveal an obvious error. Although the variable simplex algorithm outperformed the genetic algorithm on the Colville response surface in terms of the number of iterations needed in order to obtain convergence, the dependence of the convergence on the initial values was considered enough reason not to investigate the use of the variable simplex in the catchment optimisation problem. Early attempts at using the variable simplex algorithm on the catchment problem showed that there was no convergence in control parameter values after 400 iterations. Consequently, despite its computational expense, it was considered necessary to use the genetic algorithm approach for the catchment optimisation problem.

The in-stream toxicity stressor values generated by the α-cut method and the corresponding effect expectation values are shown in Fig. 4.4. The first two trials involved a comparison of the choice of initialisation option with the use of the average minimum aggregation for λ_{x}.

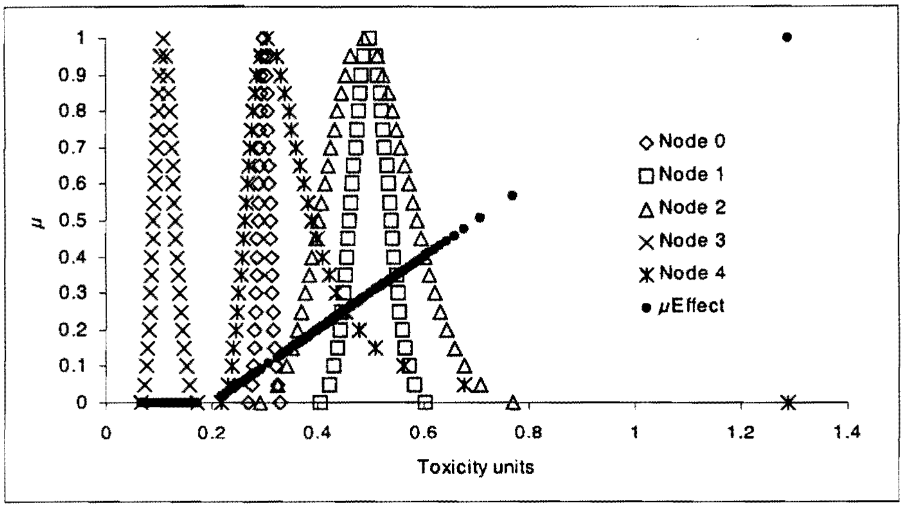


Figure 4.4. The possibility distribution of toxics as calculated at each node before attenuation. By way of comparison the toxic effect membership values used in the calculation are also plotted.

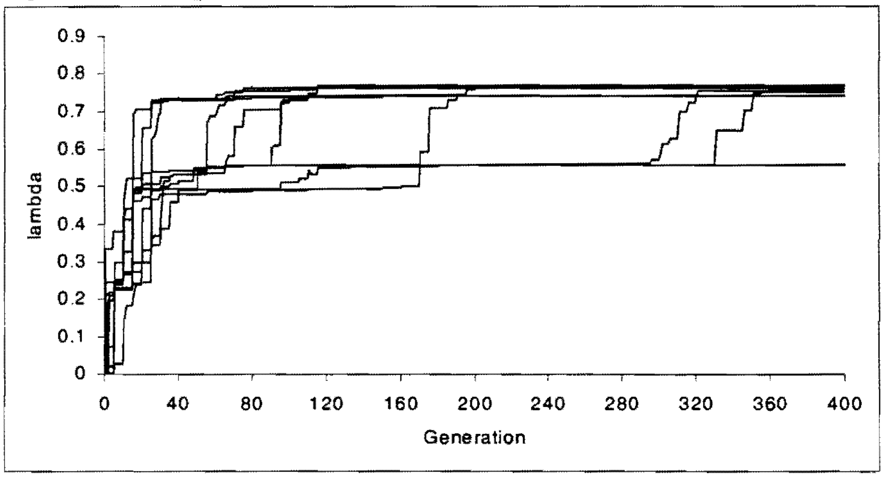


Figure 4.5. The best λ as function of the number of generations per cycle with Option 1 using the average minimum aggregation for λ_∞.

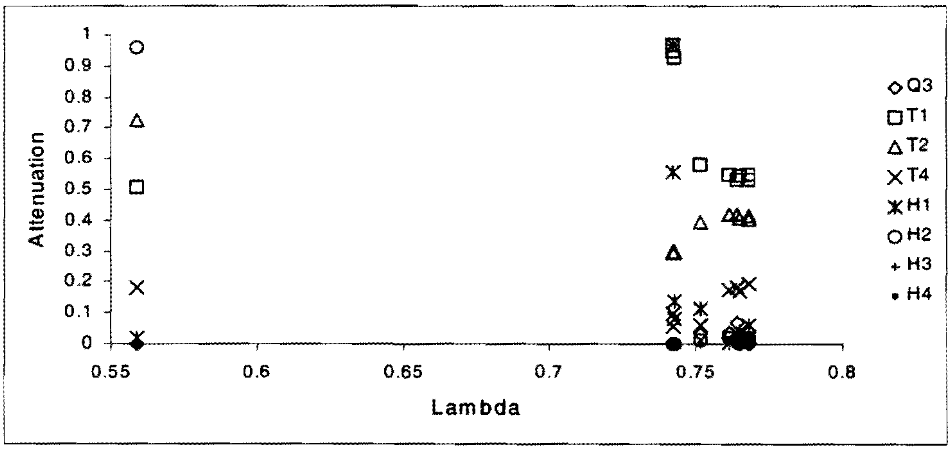


Figure 4.6. The attenuation values corresponding to the best λ in each cycle with Option 1 using the average minimum aggregation for λ_∞.

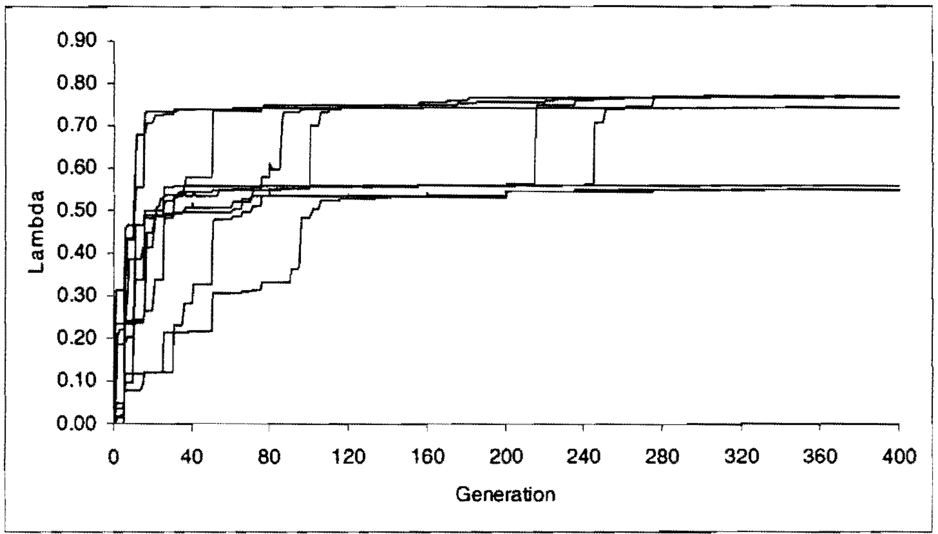


Figure 4.7. The best λ as function of the number of generations per cycle with Option 2 using the average minimum aggregation for λ_{x}.

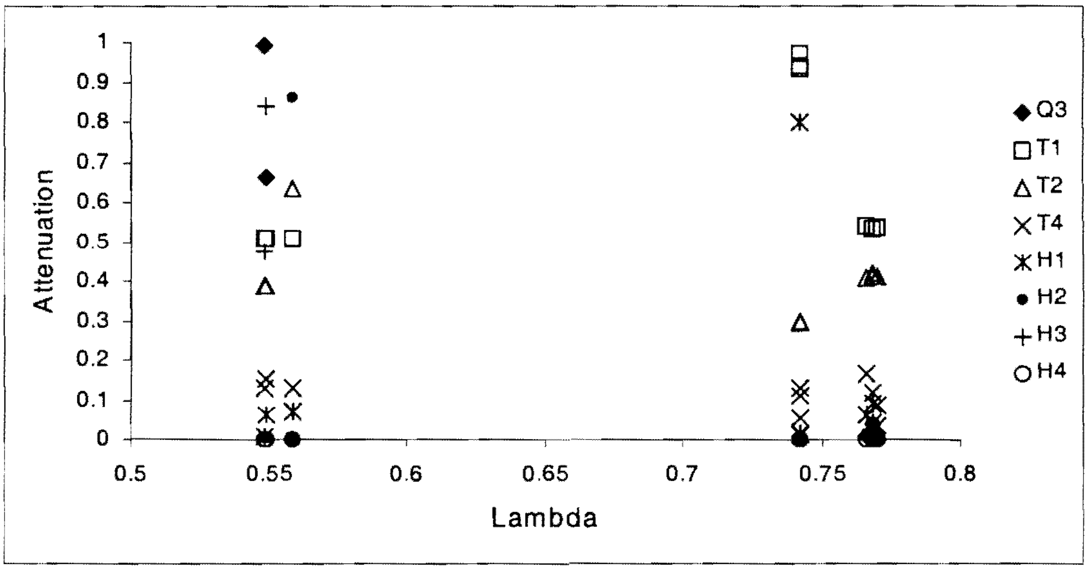


Figure 4.8. The attenuation values corresponding to the best λ in each cycle with Option 2 using the average minimum aggregation for λ_{x}.

The convergence rates of λ are compared in Figs. 4.5 and 4.7. Both Figures show that there are probably two minima: one with λ = 0.72 and the other with λ = 0.54. Option 2 (both population initialisations from exponential distributions) shows a marginally better convergence rate then Option 1. Comparison of Figs. 4.6 and 4.8 shows the optimal attenuation vectors for the two options compare well.

Toxicity attenuation requires the most attention, as can be expected from the possibility distributions, with source 1 requiring the highest attenuation. This corresponds well, with the intuitive notion that the relatively high toxicity and habitat degradation values at node 1 will result in an increased overall risk just downstream of node 1. The flow and habitat stressors need little attenuation ($x_{i,j}$ < 10%).

The attenuation values in Figs. 4.6 and 4.8 show discrimination among identical stressors (e.g. toxics) as well as raising the issue of neglect of specific source satisfaction. Here, average minimum aggregation may well balance a zero satisfaction at one source with a higher satisfaction at another source. This might argue for applying minimum satisfaction aggregation of individual stressor satisfaction.

When both minimum satisfaction aggregation and equity constraints are applied to the Option 2 algorithm, the results in Figs. 4.9 and 4.10 are obtained. This shows that the convergence rate of the algorithm has slowed down significantly so that in 400 generation the best satisfaction,λ, was only about 0.15. The stressor attenuation appears satisfactory from an equity point of view but it was attained at the cost of higher flow-stressor attenuation.

The lower overall λ might suggest that this application places an unfair burden on stressor sources. The question is if the imposition of risk constraints is the cause of the lower λ. Comparison of Figs. 4.11 and 4.12 with Figs. 4.6 and 4.9 would suggest that λ be dominated by λ_x. Other data (shown in Appendix 4) indicated that the risk satisfaction level, λ_r, is highly variable but in the runs corresponding to Figs 4.11 and 4.12, $\lambda_r \in$ [0.78, 0.99] and $\lambda_r \in$ [0.16, 0.99] respectively. This would seem to indicate that while the risk constraints might steer the control variable selection in the direction of lowest λ_x.

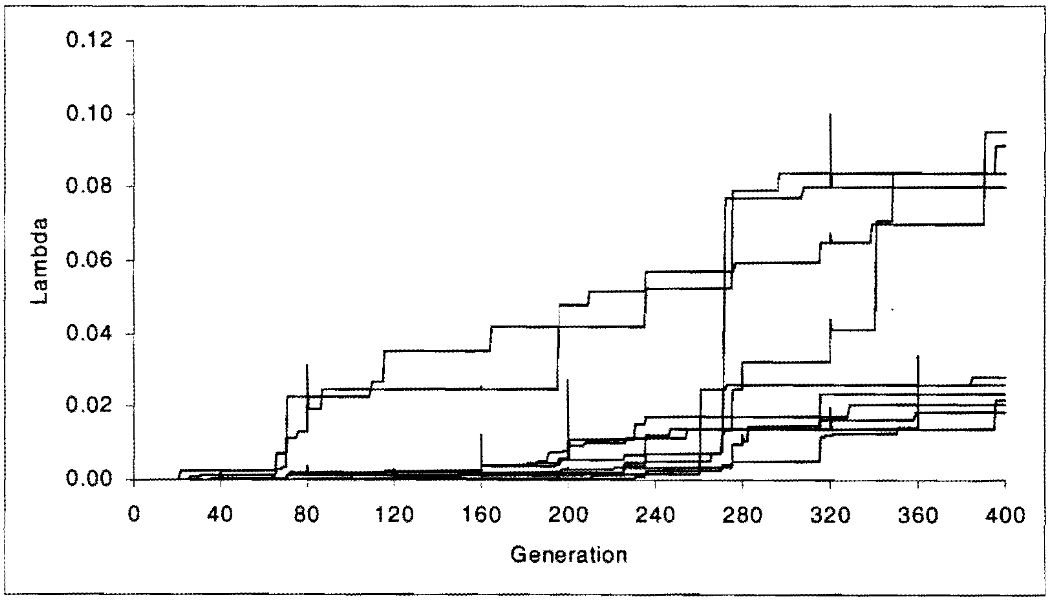


Figure 4.9. The best λ as a function of number of generations in a cycle with Option 2 and including disjunctive aggregation for λ_x and stressor specific equity constraints.

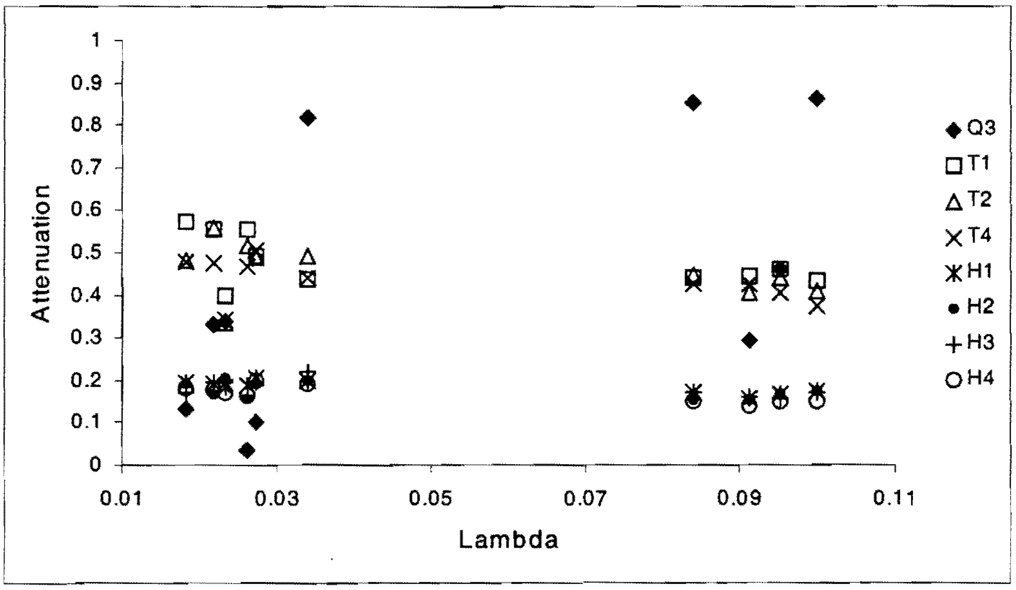


Figure 4.10. The attenuation values corresponding to the best λ per cycle with Option 2 and including disjunctive aggregation for λ_x and stressor specific equity constraints.

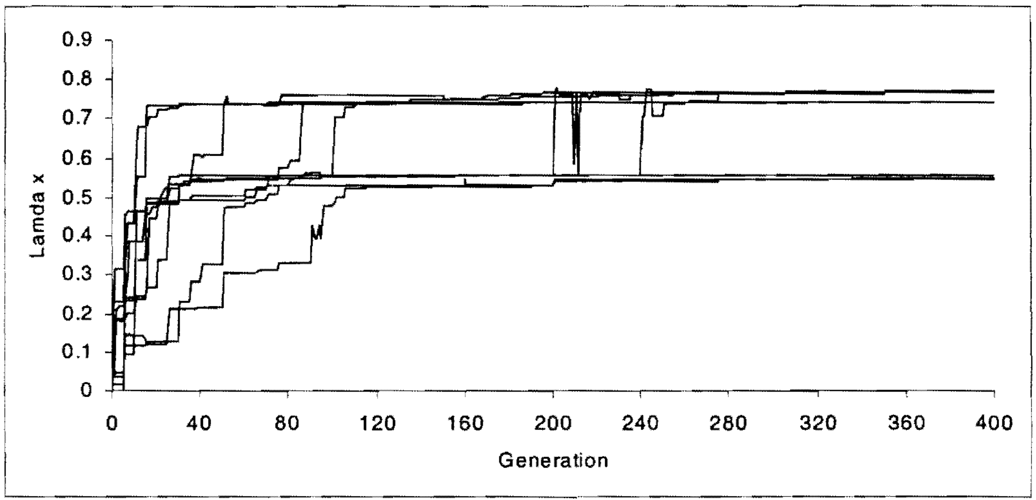


Figure 4.11. Attenuation satisfaction λ_x as a function of the number of generations with Option 2 and average minimum aggregation for λx (no equity constraints). Comparison with Figure 4.7 shows that λ is dominated by λ_x.

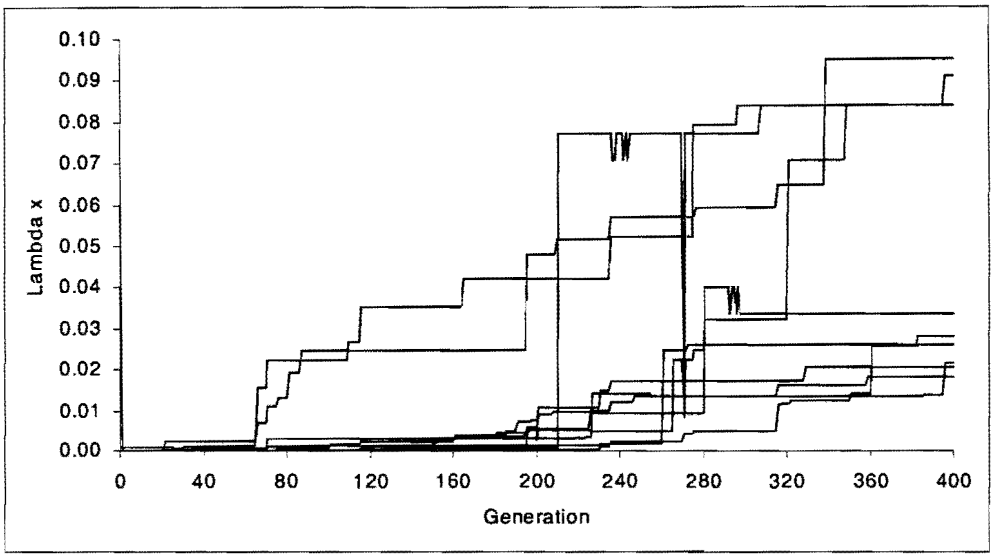


Figure 4.12. Attenuation satisfaction λ_x as a function of the number of generations with Option 2 with disjunctive aggregation for λx and stressor attenuation equity constraints. Comparison with Figure 4.9 shows that λ is dominated by λ_x.

The stressor attenuation values predicted by this algorithm are listed in Table 4.5.

Table 4.5. Stressor attenuation values for various algorithm options.

	Option 1	**Option 2**			
Equity constraint	No	No	No	Yes	Yes
λ_x aggregation	Av. Min.	Av. Min.	Conjunctive	Av. Min.	Conjunctive
x_{T1}	0.039	0.004	0.141 – 0.914*	0.289	0.461
x_{T2}	0.549	0.563	0.689 – 0.957*	0.497	0.461
x_{T4}	0.404	0.410	0.542 – 0.993*	0.522	0.440
x_{Q3}	0.062	0.086	0.023 – 0.167*	0.515	0.405
x_{H1}	0.060	0.037	0.047 – 0.833*	0.071	0.169
x_{H2}	0.004	0.160	0.067 – 0.915*	0.068	0.168
x_{H3}	0	0	0.043 – 0.964*	0.070	0.159
x_{H4}	0	0	0	0.074	0.149

* Variable attenuation values with a degenerate $\lambda = 0.99$

The computation time for this optimisation could be significant. An optimisation code written in Microsoft® QBASIC (in which the development was done) running on a 333 MHz Pentium II processor took between about 3 hours to complete the optimisation. While it is recognised that substantial computation time saving can be brought about by more efficient coding, computation time is likely to remain significant. However, in comparison to the time required to perform stochastic WLA's, this time expenditure is probably not excessive.

4.4.5 DISCUSSION

Determining the source specific stressor-attenuation values by the optimisation of ecological concern to process-related acceptability appears a viable method to arrive at site or situation specific management criteria.

In the example used above, it has tacitly been assumed that the methodology exists by which the stressor-specific response curves can be generated. In all cases, this would involve a significant amount of effort. In most cases such methodology is not readily available or is still subject to development.

In the case of toxics, recourse will likely have to be taken to ecotoxicological data. However, the common laboratory scale LC50 or EC50 data on its own, is hardly likely to suffice. The selection of the correct metric to represent the ecosystem-level effects is a subject for expert deliberation based on system specific knowledge.

In the case of flow related stress, it seems feasible that some of the developments currently under way on the estimation of in-stream flow requirements (e.g. King and Louw, 1998) could eventually be used to parameterise the flow-stress response relationship.

Habitat stress response is likely to be an expert-input driven assessment and the level of input very similar to that of a risk assessment. In fact, the input required for each stressor is virtually the same as for the effect assessment phase of an ecological risk assessment of each stressor.

While the data and information requirements of this approach are high, the potential exists for each water user (where "use" is defined not only in terms of abstraction but also as discharge) as well as the regulator to effect compromises. At the same time the water users are required to consider their requirements carefully. Although simple trapezoidal acceptability functions were used in this example, these functions could be quite complex, without detriment to the overall process.

The risk objective values clearly have a significant impact on the attenuation values estimated by this procedure (Appendix 4, Figs A4.4.5 and A4.4.6). It can make a very dramatic difference in the attenuation of toxics at source 1, with resultant cost and other implications. Careful attentions need to be given to the derivation of these values so that they correlate to field observations such as biomonitoring results.

Given the complexity of the process in deriving the information necessary to perform this optimisation, it is unlikely that this approach to stressor attenuation calculation will be used at a primary level. A typical application scenario would require that a hazard-based screening tier would precede the use of this model. As the rate of return of environmental benefits slows down when increasingly strict effluent standards are applied, a critical appreciation of effect-based models (such as the ecological concern model used here), will become increasingly important (Somlyódy, 1997). Affordability in river basin management can be addressed by the combined use of effluent criteria (as a minimum requirement) and ecological risk or concern objectives as means to refine and adapt such criteria.

CHAPTER 5

CONCLUSION: APPLICATION AND THE WAY FORWARD

5.1 ABSTRACT

This chapter describes three possible applications for the models developed in preceding chapters as tools in resource directed catchment management:

- Rapid hazard ranking in rapid reserve estimates,
- The derivation of in-stream stressor specific criteria, and
- The derivation of baseline point-source criteria in catchments under development pressure.

Some of the necessary work that needs to be performed to place risk-related catchment management on a sound scientific basis and incorporate it in the current water resource management practice include:

- Development of a policy on risk assessment and risk management
- Deriving risk objectives
- Establishing a risk communication policy
- Investigating more efficient optimisation algorithms
- Deriving and updating stressor response relationships (SRR's)
- Development of rigorous methodology for the characterising stressor attenuation acceptability.

5.2 IN SUMMARY

In Chapter 1 a potential Reserve-related problem in dealing with diverse stressors from multiple sources against the background of the mandatory protection of a sustainable aquatic ecosystem. It was proposed that it be by risk methodology. A broad outline of how ecological risk-based management (ERBM) might be applied, was given.

In Chapter 2 the ecological risk assessment (ERA) methodology was outlined and how it needs to be adapted for ERBM. Considering its theoretical background it was clear that the problem of projecting end-points from laboratory scale data to the ecosystem level involves a large amount of uncertainty since it requires not only scale projection but also conceptual projection. This process needs to be performed for each individual stressor risk. The end-point projection forms a very important task in the construction of SRR's.

Chapter 3 modelled the aggregate risk of diverse stressors as the disjunction of individual stressor risk. It was illustrated how this type of aggregation could be used in both a probabilistic and possibilistic framework.

Chapter 4 modelled the diverse-stressor-multiple-source problem as an optimisation problem. A genetic algorithm was chosen to solve the optimisation problem, not because it is necessarily the most efficient, but because it is conceptionally simple. It was illustrated that it is indeed possible to obtain source- and stressor-specific attenuation criteria.

5.3 A PERSPECTIVE ON THE WORK PRESENTED

It became clear that the inputs needed to make these procedures functional are quite information- and knowledge-intensive. Even though the necessary knowledge exists, the risk-based decision-making is unlikely to be a first choice approach unless the stakes are high enough to warrant the time and effort to generate the necessary data.

5.3.1 INFORMATION NEEDS

The scientific input to risk methods largely comprises of uncertainty, variability and vagueness characterisation as well as risk characterisation. All of these depend strongly on insight into the functioning of the aquatic ecosystems, as expressed in conceptual models of various kinds, the type and quality of data available and the experience and insights of the body of experts available in the country. When moving towards the benefits of using risk-based methodology, it should be recognised that the quality of scientific basis of risk-techniques need to be carefully considered and expertise in a small country, like South Africa, needs to be nurtured.

5.3.2 ACCOMODATING UNCERTIANTY

ADJUSTMENT IN THE REGULATORY PARADIGM

The regulatory mechanisms need to be adjusted to (a) recognise that uncertainty (in it's broadest sense) is a fact of life in ecosystems management and (b) that rather than to try to define the uncertainty out of the process, incorporate the methodology to deal with it in the process. A vast literature exists in the area of business and engineering decision-making under uncertainty (see for example Chapter 1 in Stewart, *et al*, 1997), so that uncertainty need not be seen as a bane to regulatory decision-making.

RISK COMMUNICATION NEED

By nature, human beings have a fear of the unknown and of uncertainty. Innately, therefore, when a decision is made in an area of which they do not have knowledge and by mechanisms they do not understand, people tend to be distrustful. If, in addition, they suspect that the motives behind the decision are suspicious or antagonistic to their value system, distrust may turn to hostility.

Suspicion of the scientific domain may lead to remembering catastrophes of the past, such as the thalidomide scandal of the sixties and the uncontrolled use of DDT in the 1950's. The use of risk by the scientific community and particularly in the industrial context has been seen as an excuse for doing nothing (Tal, 1997). These issues need to be addressed by effective risk communication, which is generally recognised as an increasingly important aspect of risk application (CRARM, 1997; OECD, 1997; Yosie, et al., 1998).

5.4 POSSIBLE APPLICATIONS OF THE METHODOLOGY DESCRIBED IN THIS STUDY

Risk may reasonably be used to aid water resource quality management decisions and activities related, but not necessarily limited to the following areas:

5.4.1 BASIS FOR STRESSOR-SPECIFIC WATER QUALITY CRITERIA.

The current South African Water Quality Guidelines for the Aquatic Environment has been derived from toxicological data (typically concentration-response data) and some qualitative assumptions regarding exposure. These criteria have the following limitations:

- the derivation process produces anomalous risk results so that the expected effect differs from substance to substance,
- recognition could not be given to the co-occurrence of different stressors since they could not be expressed on a common basis, and
- the criteria do not necessarily relate to the same ecological effect.

Redefining and recalculating the criteria on a risk basis induces a measure of transparency into the interpretation of the criteria. The other criteria (besides those for the aquatic environment) could be approached similarly. Both ERA and human health risk assessment will be important here. Methodologies have been developed for the determination of the ecological reserve. These methodologies follow relatively independent routes to establish stressor-specific management criteria. These criteria characterise the reserve for a particular river reach. In the form these criteria are currently expressed, there is no description of the uncertainty component in the relationship between the stressor and its effect. It is likely that the various stressor criteria project to different risk levels. A significant improvement in the homogeneity of the process can be brought about by:

1. describing the management classes in risk terms
2. adopting suitable numeric risk objectives
3. deriving SRR's for effect likelihood at the statutory end-point for all identified stressors
4. adopting numeric risk objectives which are related to the management goal
5. calculating the corresponding stressor exposure-likelihood level and hence the management criteria for the designated stressors by iterative application of the models in Chapter 3.

Each of the steps 3 and 5 above can be performed at various levels of environmental realism, ranging from a highly simplified desktop estimate, which is a rapid, low confidence, estimate to a moderately long term, high confidence site-specific study.

In its simplest form this procedure would involve:

(a) Assuming a type distribution for the stressor (e.g. a lognormal distribution).

(b) Iteratively adjusting the location and scale parameters of the distribution and comparing the calculated risk from each parameter vector with the risk objective. This would call for optimisation and may involve two dimensional uncertainty analyses.

(c) Describing benchmarks of the stressor distribution (e.g. median and 95th percentile).

It is clear that the quality of the SRR is vitally important.

5.4.2 THE DERIVATION OF BASELINE POINT-SOURCE CRITERIA IN CATCHMENTS UNDER DEVELOPMENT PRESSURE.

The issue of diverse stressor-multiple source management under constraints was the main focus of this study. The technical process of the multiple-source problem is described in Chapter 4. The diverse-stressor problem formulation requires some extra information. The first four steps of 5.4.1 is followed, but the following steps are added:

5. define catchments or river reaches subject to development pressure,
6. obtain source- and stressor-specific upper and lower limits of stressor attenuation from stressor sources with particular attention to the uncertainty in these estimates,
7. define, either as a matter of policy, or pragmatically, the relative weighting of source and regulator satisfaction,
8. estimate the source attenuation terms along with its confidence estimates, and
9. finalise the management criteria by negotiation between regulator and regulatee(s) based on attenuation estimates.

The derivation of the stressor-source specific attenuation must be followed by a calculation of the actual stressor values represented by the level of attenuation. This could then be compared to the source criteria derived from WLA for example (in the case of substance stressors). In evaluating the implications of different Hazard- or risk-based in-stream stressor criteria and the criteria derived in terms the DSMS solution it should be remembered that:

- the DSMS criteria are risk based and therefore not comparable to hazard-based criteria
- the DSMS criteria are derived from catchment considerations and do not address site-specific considerations.

If the DSMS stressor criteria are more lenient than the other criteria, the DSMS criteria might serve as the short-term criteria but with the proviso that whichever constraints hamper the achievement of the other criteria should be resolved on the longer tem. If the converse is true, the stricter of the two should be used.

5.4.3 RESOURCE MANAGEMENT CLASSIFICATION.

The provision in the National Water Act for the classification of water resources can reasonably be linked to risk concepts. Management objectives may more specifically be expressed in terms of allowable risk to the Reserve. This provides an explicit communality between the receiving water quality/risk objectives and the Reserve as well as effluent criteria and/or standards.

5.4.4 HAZARD RANKING.

In some situations, it is neither necessary nor feasible to calculate absolute risks. In the case where different hazards within the same scenario or hazards in different scenarios need to be compared, risk is often a suitable basis for comparison. The management criteria derived in the current reserve determinations (McKay, 1999) are largely hazard based. Realistic ranking of the hazards addressed in this process can be accomplished by estimating the risk attached to these hazards. This would require:

- ⇨ a clear statement of a realistic worst case stressor exposure scenario,
- ⇨ a clear conceptual ecological model linking the level of data with the required end-point,
- ⇨ an expression of the uncertainty in the SRR, and
- ⇨ an estimate of the risk.

This will aid in characterising the uncertainty and channelling expenditure into areas of greatest return.

5.5 ISSUES IN THE APPLICATION OF RISK METHODOLOGY

The major areas where attention needs to be given to give effect to risk-based catchment management are:

5.5.1 THE DEVELOPMENT OF A POLICY ON RISK ASSESSMENT AND MANAGEMENT

Some aspects involved in a policy on risk and risk assessment include:

- A common understanding of the definition of risk.
- How risk is seen in relation to other paradigms.
- What conditions might indicate the use of risk methodology
- Adoption of a tiered approach to the use of risk as an assessment technique
- Minimum requirements for risk assessment.

An analysis of the regulatory situation in other countries (Table 5.1) shows that the lack of a legal basis for the explicit use of risk methodology in South Africa is not unique. The National Water Act (like many other laws in South Africa) allow for the promulgation of regulations under the Act and application of risk may well be described in such regulation.

Table 5.1 An assessment of legal standing of risk assessment in selected countries (based on OECD, 1997).

Country	Law prescriptive/ goal setting	Risk criteria identified/ specified	Quantified risk assessment recognised
Germany	Prescriptive	No	No
France	Some prescriptive	Yes (zoning)	No
Switzerland	Both	In guidelines	Yes
UK	Both (more goals)	In guidelines	Yes
USA	Goal	Specific goals and definitions	No (can be used)
Norway	Goal (by industry)	No	Yes (implicitly)
Netherlands	Goal	Yes (not in law yet)	Yes

The OECD (1997) notes a potential legal problem in explicitly incorporating risk in laws since it may be asked whether generating and accepting a measure of risk will infringe the rights of individuals. This will clearly have to be assessed on a country-specific basis.

5.5.2 DEVELOPING RISK OBJECTIVES

In the foregoing work, it had been implicitly accepted that recognised risk criterion values are available, whether crisp or fuzzy. Such values for aquatic ecosystems are rare if existing at all. The reason, most likely, is that consensus on the actual numeric value as well as the descriptive risk, is likely to depend on the specific situation that is being assessed and factors such as the protection value of the ecosystem will probably have an impact. The situation with the ecological Reserve in South Africa already lends itself to a discretisation of aquatic ecosystems. An importance and sensitivity rating of river systems is being developed for river reaches (Kleynhans, 1999a), which will be factored into the Reserve determination. This could serve as a basis for ascribing maximum acceptable risk values depending on the importance class.

The decision on numeric risk criteria, i.e. what levels of probabilistic and possibilistic risk are considered acceptable, for human health considerations are generally founded on those used by the USEPA. For carcinogens a risk limit of 10^{-6} per lifetime is accepted and for non-carcinogens a value of 10^{-4} per lifetime.

For ecosystems the acceptable risk limit is likely to be more problematic. The values that will be accepted may well depend on the end-point. The risk of a major fish-kill and that of long term unsustainability may be perceived differently because the end-point relate to different time-scales. A fish kill may, because of the immediacy of effect, be rated higher than a long-term effect.

A recent study (Jooste *et al.*, 2000) considered the setting of risk objectives (RO's) by comparison with actuarial risk values. Some of the suggested values are listed in Table 5.2. These could be combined with the qualitative description in Table 5.3 to provide probabilistic risk criterion values.

Table 5.2 Human mortality risk benchmarks for establishing and communicating risk (from Chapman and Morrison, 1994)

Cause	Probability
Motor vehicle accident (USA)	1 : 100
Smoking (20/day) all effects	1 : 200
Murder	1 : 300
Fire	1 : 800
Firearm accident	1 : 2 500
Electrocution	1 : 5 000
Asteroid/ comet impact	1 : 20 000
Passenger aircraft crash	1 : 20 000
Flood	1 : 30 000
Tornado	1 : 60 000
Venomous bite/ sting	1 : 100 000
Fireworks accident	1 : 1 000 000
Food poisoning (botulism)	1 : 3 000 000
Drinking water with EPA limit of trichloro-ethylene	1 : 10 000 000

Table 5.3 A semi-quantitative approach to risk characterisation

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

The occurrence of some of the ecological events described in Table 5.2 may be difficult to define. It may, for example, be argued that smoking constitutes a generally acknowledged high risk activity and that, therefore, the highest risk that will be allowed for a chosen significant end-point will also be 1: 200. On the other hand, flying in a passenger aircraft is generally considered safe and that, therefore, a risk of 1: 20 000 may be considered negligible. These values would likely be determined on a case specific basis.

5.5.3 RISK COMMUNICATION

In the catchment management situation, which is also the likely setting for the diverse-stressor-multiple-source problem, it could be envisaged that communicating and defending the risk criteria selected for a river reach would arise. This requires dealing with the sociological problem of risk perception. Perceptions about risk change with changing circumstance and increasing familiarity; increased familiarity with a hazard leads to a better estimate of its true probability of occurrence, or conversely, the more unfamiliar one is with a hazard, the more one is inclined to overestimate the danger (OECD, 1997; Tal, 1997). The way in which risks are communicated in a tense situation, could have a significant impact on the viability of the methodology described in Chapter 4 particularly.

5.5.4 INSITUTIONALISING RISK MANAGEMENT IN THE RESERVE CONTEXT

There needs to be a formal awareness of uncertainty in ecological management. This would involve an institutional concern with the variability, uncertainty and vagueness pertaining to the ecosystem and an insistence on all management levels of explicitly stating or asking for such expressions, in order to contextualise management decisions. This would involve:

- Developing a generic "first attempt" ecological model for risk assessment.
- Cultivating an institutional awareness of SRR's and their importance in effect driven management
- Creating risk-susceptible administrative procedures e.g. risk oriented discharge permits
- Developing risk assessment capacity
- Developing risk communication capabilities

5.6 RESEARCH NEEDS: THE WAY FORWARD

The work presented in this study on the derivation of effect-likelihood criteria in a diverse-stressor multiple-source (DSMS) management situation, addressed an aspect of ERA that had not received much attention in the past. Some of the issues addressed in this study require a multi-disciplinary or trans-disciplinary approach, which increased the difficulty of the task significantly. Some of the issues were, consequently, left unresolved although they may be quite significant. Some of the more significant problems that would still need to solved include:

1) Investigating the use of other optimisation algorithms, e.g. simulated annealing and stochastic optimisation methods. The genetic algorithm that was used in the DSMS problem solution,

although sufficient for the small number of control variables in the illustrative situation used, may not work as well in a higher dimensional space.

2) Deriving stressor-response relationships for all common stressors to reserve related end-points. The possibilistic approach used in Chapters 3 and 4 may not suffice in situations where higher precision values are necessary. The probabilistic analogue to this approach needs to be researched.
3) Establish formal feedback loops between SRR's and instream bio monitoring to inform and improve both the SRR's and the biomonitoring programme design. Once again, the possibilistic Dempster-Schafer approach using possibility distributions has to be extended to the probabilistic analogue. This may involve investigating the use of Bayesian methodology.
4) Improving the stressor modelling sophistication of the model in Chapter 4. The Possibilistic approach was chosen because it appeared that the data were better suited to the situation. Both the stochastic approach and a more sophisticated environmental model could be used to improve the realism of the stressor value prediction in suitable situations.
5) Developing methods to characterise source attenuation acceptability in a rigorous manner. The assumption in Chapter 4 had been that suitable methodologies exist by which stressor source managers could estimate the acceptability of attenuation values. It is not immediately apparent that these methods already exist and some effort might well required to formulate credible, transparent methodology to define such acceptability values rigorously.

APPENDIX TO CHAPTER 1

A1.1 A REVIEW OF SOME PERTINENT ASPECTS OF THE SOUTH AFRICAN NATIONAL WATER ACT (ACT 36 OF 1998).

The aim of this study is to provide a tool to be used in water resource management with a view to the protection of the aquatic ecological Reserve as defined in the National Water Act in South Africa. While the application of the approach may be much wider than the aquatic system, this study must be seen against this backdrop.

In its preamble, the rationale for the Act comes from recognising that:

(a) "water is a scarce and unevenly distributed resource",

(b) "the ultimate aim of water resource management is to achieve the sustainable use of water for the benefit of all users",

(c) "the protection of the quality of water resources is necessary to ensure sustainability of the nation's water resources" and

(d) there is a "need for the integrated management of all aspects of water resources".

Section 2 of the Act states that 'the purpose of this Act is to ensure that the nation's water resources are protected, used, developed, conserved, managed and controlled in ways which take into account amongst other factors-

(a) meeting the basic human needs of present and future generations;

(b) promoting equitable access to water;

(c) ...

(d) promoting the efficient, sustainable and beneficial use of water in the public interest;

(e) facilitating social and economic development;

(f) ...;

(g) protecting aquatic and associated ecosystems and their biological diversity

(h) reducing and preventing pollution and degradation of water resources; ...

Some of the pertinent definitions that will be used here will be used in a manner similar that in the Act:

" ....

(iii) '**catchment**' in relation to a water course .... means the area from which any rainfall will drain into the watercourse.... Through surface flow to a common point or points.

(xi) '**in stream habitat**' includes the physical structure of the watercourse and the associated vegetation in relation to the bed of the watercourse;

(xv) '**pollution**' means the direct or indirect alteration of the physical, chemical or biological properties of the water so as to make it- ....;

(b) harmful or potentially harmful-

(aa) to the welfare health or safety of human beings;

(bb) to any aquatic or non-aquatic organisms;

(cc) to the resource quality; or ...;

(xvii) '**protection**' in relation to a water resource, means- (a) maintenance of the quality of the water resource to the extent that the water resource may be used in an ecologically sustainable way; (b) prevention of the degradation of the water resource; and (c) rehabilitation of the water resource;

(xviii) '**Reserve**' means the quantity and quality of water required-

(a) to satisfy basic human needs....; and

(b) to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource

(xix) '**resource quality**' means the quality of all aspects of a water resource, including-

(a) the quantity, pattern, timing, water level and assurance of in stream flow;

(b) the water quality , including the physical, chemical and biological characteristics of the water;

(c) the characteristics and condition of the in stream and riparian habitat; and

(d) the characteristics, condition and distribution of aquatic biota;

(xxii) '**waste**' includes any...material that is suspended, dissolved or transported in water (including sediment) and which is ... deposited ... into a water resource in such volume, composition or manner as to cause ... the water resource to be polluted;

(xxiv) '**watercourse**' means ... a river ... [or] a natural channel in which water flows regularly or intermittently ... and ... includes, where relevant, its bed and banks;

(xxvii) '**water resource**' includes [*inter alia*] watercourse [and] surface water.

......".

Section 6 of the Acts requires that the water resource strategy (which may be phased) should (6 (b) (i)) provide for the requirements of the Reserve and (6 (i)) state the water quality objectives for the water resource.

Sections 12 and 13 make provision for the classification of the water resource, although it does not specify the basis for classification. This classification system must also serve as the basis for setting the resource water quality objectives. The objectives may relate to:

(a) the Reserve
(b) the in stream flow
(c) the water level
(d) the presence and concentration of particular substances in water
(e) the characteristics and quality of the water resource and the in stream and riparian habitat
(f) the characteristics and distribution of aquatic biota
(g) the regulation of in stream or land-based activities
(h) any other characteristics

of the water resource.

The impact of the Reserve on water use and water management can be seen by considering that:

- Section 15 makes it mandatory that any action that follows from the Act must give effect to this class and its associated water resource quality objectives while Section 18 demands that such actions must also give effect to the Reserve. Section 16 determines that the Reserve must also be set in accordance with the class. This places the Reserve central to water resource management.

- Under Section 22. (7)(b)(i) compensation which is payable on the reduction of lawful use of water does not apply to reduction of water use to make provision for the Reserve.

- Section 56 makes provision for establishing a pricing strategy which may contain a strategy for water use charges for funding water resource management to protect the resource, including the discharge of waste and the protection of the Reserve (55.(2)(a)(iv)).

In making regulations on water use, besides giving effect to the Reserve and the resource classification system, Section 26 requires that, *inter alia*, consideration be given to promoting economic and sustainable use of water and to conserve and protect the water resource and the in stream and riparian habitat. Water use regulation must take into account factors such as (Section 27. (1)):

1. The socio-economic impact of water use or curtailment of use (d)
2. The catchment management strategy applicable to the resource (e)
3. The likely effect of the water use on the resource and other users (f)
4. The class and resource quality objectives (g)
5. The investment already made and to be made by the water user (h)
6. The quality needs of the Reserve and to meet international obligations (j)

A1.2 RISK AND HAZARD: PARADIGMS AND STYLES

A1.2.1 THE HAZARD AND RISK ASSESSMENT PARADIGMS

Given that monitoring and assessment are essential components of any management strategy, the assessment paradigm is crucial to the expectations and format of the assessment of management goal attainment. The assessment may take the form of either a quantal or a continuous metric. The quantal assessment paradigm (QAP) and continuous assessment paradigm (CAP) are referred to as hazard and risk assessment paradigms (Figure A1.1) respectively by Suter, (1993). The characteristics of these paradigms are summarized in Table A1.1 and the progress of an assessment according to these paradigms is illustrated in Figure A1.1.

Table A.1.1. Characteristics of environmental hazard assessments and risk assessments (adapted from Suter, 1993). Some of the characteristics are explained in the text.

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

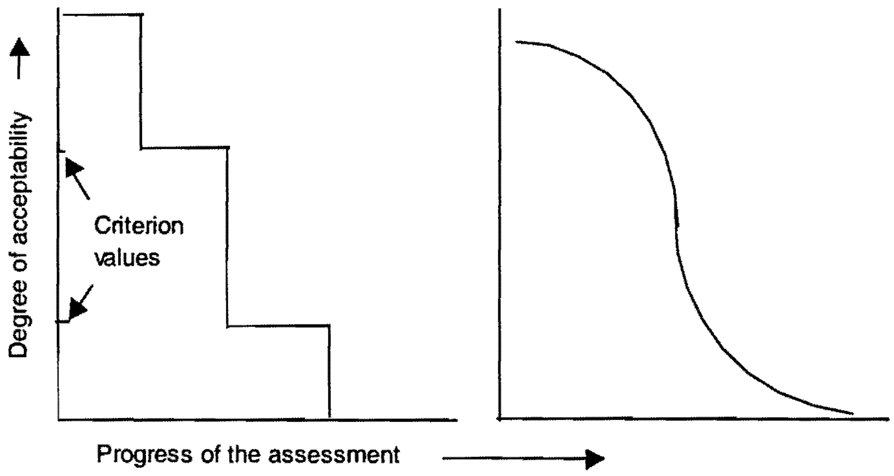


Figure A1.1 A representation of the outcome of an assessment as the assessment progresses. In the progress of the assessment, the confidence in the data increases. In this example both assessments starts out with the assumption of unacceptable for a situation that is essentially acceptable.

A comparison between the QAP and CAP paradigms reveals:

1. Both QAP and CAP assume that the environmental safety of a substance should be based on the relationship between the degree of toxicity and the extent of exposure. This differs in principle from technology-based assessment.

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

a) Reliance on scientific **judgement** or "expert opinion" of what constitutes "acceptable" or "unacceptable". The expert opinion may be either explicitly stated or encapsulated in a criterion value (CV).

b) ANOVA techniques and statistical **hypothesis testing** play an important role in the QAP in deciding whether the expected (or measured) environmental concentration (EC) differs from the CV.

c) A fundamental assumption of the QAP is that, given enough time and effort, the situation where the EC, for example, cannot be confidently fit into either category, can be resolved (i.e. it can in principle always be assigned a unique outcome). In a situation where no clear, unequivocal answer is available in assessing the status of an observation relative to the criterion, the hazard paradigm demands tiered iterative data gathering (testing and measurement) procedure until a definitive answer can be given. This gives rise to a **tiered assessment**. As more iterations are added to the process the confidence in the distinction between acceptability and unacceptability grows. Confidence here does not necessarily refer to statistical confidence, but more so to institutional or personal confidence (Suter, 1990).

d) Formally, there is not necessarily an explicit decision *ab initio* as to which end-points that are being addressed; it does not intend to identify what is specifically expected to occur (Bartell, *et al*, 1992) since these are implicit in the criteria. Both the process by which the expert selects the end-point (i.e. what might be expected to occur) and the extent to which this is possible is subjective to a degree even though it may be internally coherent. This aspect of the QAP makes the process inherently less transparent.

3. The continuous assessment paradigm (CAP) is characterised by:

a) Acceptance, *a priori,* that some uncertainties are practically irreducible and that a definite decision on yielding acceptable/unacceptable may be logically impossible. Consequently, there are decisions that may never (within the time frame of the decision making process) have a deterministic answer and therefore relies more heavily on **probabilistic expression**.

b) Accepting a **continuum** "grey scale" in assessment outcome. This results from its use of probabilistic assessment methods to accommodate uncertainty explicitly.

c) Because of its probabilistic expression, the object and **end-point appears explicitly** in the assessment (the probability of what could happen to whom).

d) In most environmental assessment situations, the risk paradigm would appear to be **more objective** means of decision-making. It must however be accepted that some form of human judgement can never be completely removed from the risk paradigm. For example, what constitutes a large or a small risk is often a matter of subjective judgement or policy.

APPENDIX TO CHAPTER 2

A2.10 ASPECTS OF END-POINT PROJECTION

A2.10.1 THE CRISP INFERENTIAL RULE BASE

The rule base deriving from the conceptual model can be stated as:

IFF *Sustainability* IS *assured* THEN *Integrity* IS *intact* **Rule I**

IF *Integrity* IS *intact* THEN (*Biodiversity* IS *adequate* AND *Temporal stress and recovery patterns* IS *largely undisturbed* AND *Biotic stress* IS *insignificant*) **Rule II**

IF *Biodiversity* IS *adequate* THEN (*Composition* IS *intact* AND *Structure* IS *intact* AND *Function* IS *normal*) **Rule III**

IFF *Composition* IS *intact* THEN NOT (*Composition stress* IS *present*) **Rule IVa**

IF (*Composition stress* IS *present*) THEN (*exposure to stressor 1* IS *present*) OR (*exposure to stressor 2* IS *present*) OR … **Dummy Rule 1**

IF (*exposure to stressor 1* IS *present*) THEN [(*significant level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *long*)] OR [(*High level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *significant*)] **Dummy Rule 2**

Combining Dummy Rules 1 and 2:

IF (*Composition stress* IS *present*) THEN {[(*significant level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *long*)]

OR [(*High level of stressor 1* IS *present*) AND (*exposure duration to stressor 1* IS *significant*)]}

OR {[(*significant level of stressor 2* IS *present*) AND (*exposure duration to stressor 2* IS *long*)]

OR [(*High level of stressor 2* IS *present*) AND (*exposure duration to stressor 2* IS *significant*)]}

OR …. **Rule Va**

(IFF denotes "if and only if")

Rules IVa and **Va** is repeated for *Structure* and *Function* to yield the equivalent rules **IVb, Vb, Vc** and **Vc** respectively. Using the key:
Sus: Sustainability is assured, *Res*: Resilience is assured, *Int* : Integrity is assured, *Div*: Biodiversity is intact, *Tpat*: Temporal stress/recovery patterns are undisturbed, *Cmp*: System composition is undisturbed, *Str*: System structure is undisturbed, *Fct*: System function is normal, *Tpats*: Temporal stress/ recovery patterns are in a state of stress, *Cmps*: System composition is under stress, *Strs*: System structure is under stress, *Fcts*: System function is under stress, *lxi0*: Minimally significant level of stressor *X* exists for integrity component *i*, *dxi0*: Minimally significant duration of exposure to stressor *X* exists for integrity component *i*, *dxi* : Long duration of exposure to stressor *X* exists for integrity component *i*, *lxi*: Intense exposure to stressor *X* exists for integrity component *i*, where $X \in$ {toxic substances (*T*), flow deficiency (*Q*), nutrient disruption (*N*), system driving variables disruption (*S*), physical habitat disruption (*H*)}, and $i \in$ {*Cmp* (*c*), *Fct* (*f*), *Str* (*s*), *Tpat* (*t*)}.

The rules can be translated to a canonical form with the standard logic operators ($\rightarrow$ "implies",$\leftrightarrow$ "equivalent to", $\neg$ "not" $\wedge$ "disjunction", $\vee$ "conjunction"):

Rules I	$Su \leftrightarrow Int$	[A2.1](Assumption)
Rule II	$Int \rightarrow Div \wedge Tpat \wedge B$	[A2.2]
Rule III	$Div \rightarrow Cmp \wedge Str \wedge Fct$	[A2.3]
Rule IVa	$Cmp \rightarrow \neg Cmps$	[A2.4a]
Rule IVb	$Str \rightarrow \neg Strs$	[A2.4b]
Rule IVc	$Fct \rightarrow \neg Fcts$	[A2.4c]
Rule IVd	$\mathrm{Tpat} \rightarrow \neg \mathrm{Tpats}$	[A2.4d]
Rule Va	$Cmps \rightarrow \bigcup_{x \in X} (lxc0 \wedge dxc) \vee (lxc \wedge dxc0)$	[A2.5a]
Rule Vb	$Strs \rightarrow \bigcup_{x \in X} (lxs0 \wedge dxs) \vee (lxs \wedge dxs0)$	[A2.5b]
Rule Vc	$Fcts \rightarrow \bigcup_{x \in X} (lxf0 \wedge dxf) \vee (lxf \wedge dxf0)$	[A2.5c]
Rule Vd	$Tpats \rightarrow \bigcup_{x \in X} (lxt0 \wedge dxt) \vee (lxt \wedge dxt0)$	[A2.5d]

Where $\bigcup_{x \in X} \bullet$ indicates the disjunction of $\bullet$ over all the stressors.

The implication of the assumption $Sus \leftrightarrow Int$ ([A2.1]) is that epistemologically sustainability does not differ from integrity. Consequently, the uncertainty associated with each of these is similar.

Given : $\neg(A \wedge B) = \neg A \vee \neg B$ and $\neg(A \vee B) = \neg A \wedge \neg B$ and if $A \rightarrow B$ then $\neg B \rightarrow \neg A$, equations [A2.5a] to [A2.5d] become[A2.6a] to [A2.6d] respectively,

$$\neg Cmp \rightarrow \bigcup_{x \in X} (lxc0 \wedge dxc) \vee (lxc \wedge dxc0) \qquad \text{[A2.6a]}$$

$$\neg Str \rightarrow \bigcup_{x \in X} (lxs0 \wedge dxs) \vee (lxs \wedge dxs0) \qquad \text{[A2.6b]}$$

$$\neg Fct \rightarrow \bigcup_{x \in X} (lxf0 \wedge dxf) \vee (lxf \wedge dxf0) \qquad \text{[A2.6c]}$$

$$\neg Tpat \rightarrow \bigcup_{x \in X} (lxt0 \wedge dxt) \vee (lxt \wedge dxt0) \qquad \text{[A2.6d]}$$

Combining Eqs. [A2.4a] to [A2.4c], [A2.3], [A2.2] and [A2.1] yields [A2.7], [A2.8] and [A2.9].

$$\neg(Cmp \wedge Str \wedge Fct) = \neg Cmp \vee \neg Str \vee \neg Fct \rightarrow \neg Div \qquad \text{[A2.7]}$$

$$\neg(Div \wedge Tpat) = \neg Div \vee \neg Tpat \rightarrow \neg Int \qquad \text{[A2.8]}$$

$$\neg Int \leftrightarrow \neg Sus \qquad \text{[A2.9]}$$

A2.10.2 FUZZY INFERENCE RULE BASE

A restatement of the crisp rules on which the inference system depend along the lines of these principles will highlight the need for a fuzzy logic approach (the ~ indicates the fuzzy formulation):

IFF *Sustainability assurance* IS *very high* THEN *Resilience assurance* IS *very high* **Rule I~**

IFF *Resilience assurance* IS *very high* THEN *Integrity maintenance* IS *very high* **Rule II~**

IF *Integrity maintenance* IS *very high* THEN (*Biodiversity* IS *normal* AND *Temporal stress and recovery patterns* IS *natural*) **Rule III~**

IF *Biodiversity* IS *normal* THEN (*Composition* IS *pristine* AND *Structure* IS *intact* AND *Function* IS *normal*) **Rule IV~**

IFF *Composition* IS *pristine* THEN NOT (*Composition stress* IS *significant*) **Rule Va~**

IF (*Composition stress* IS *significant*) THEN (*exposure to stressor 1* IS *critical*) OR (*exposure to stressor 2* IS certainly *critical*) OR ... **Dummy Rule1~**

IF (*exposure to stressor 1* IS *critical*) THEN [(*level of stressor 1* IS *marginally significant*) AND (*exposure duration to stressor 1* IS *long*)] OR [(*Level of stressor 1* IS *high*) AND (*Exposure duration to stressor 1* IS *marginally significant*)] **Dummy Rule 2~**

IF (*Composition stress* IS *significant*) THEN {[(*Level of stressor 1* IS *at least marginally significant*) AND (*Exposure duration to stressor 1* IS *long*)] OR [(*Level of stressor 1* IS *high*) AND (*Exposure duration to stressor 1* IS *at least marginally significant*]}

OR{[(*Level of stressor 2* IS *at least marginally significant*) AND (*Exposure duration to stressor 2* IS *long*)] OR [(*Level of stressor 2* IS *high*) AND (*Exposure duration to stressor 2* IS *at least marginally significant*]}

OR .... **Rule VIa~**

A2.10.3 LOWER LEVEL PHENOMENA

At this point a connection with the integrity-related variables needs to be made with the laboratory-level or other lower-level observational data. For each stressor the situation is likely to be different. The problem is that neither structure nor function nor composition might serve as an end-point at this level. This means that the type of extrapolations referred to in Table 2.2 may have to be used.

The situation for toxic substances will be developed further by way of example. The problem now is to establish how the common type of laboratory bio-assessment data can be linked to the upper-level phenomena such as structure, function and composition. In laboratory bio-assessments the two most common end-points that can be measured are mortality (*m*) and fertility or fecundity inhibition (*r*) from acute (*a*) and chronic (*c*) toxicity tests respectively.

Inferences [2.7] and [2.8] can be calculated from the conditional probabilities:

$$P(m) = P(m|\bigcup_X aX) \cdot P(\bigcup_X aX) \text{ where X} \in \{\text{T, S, Q, H}\} \qquad \text{[A3.6]}$$

$$P(r) = P(r|\bigcup_X cX) \cdot P(\bigcup_X cX) \text{ where X} \in \{\text{T, S, Q, H, N}\} \qquad \text{[A3.7]}$$

The last term on the RHS of equations [A3.6] and [A3.7] can then be expanded by using [A3.5]. However, the probability of conjunction (or intersection in set-theoretical terms) in the RHS of [A3.5] can be simplified further if the events *aX* and *cX* are independent.

$$P\left(\bigcup_X aX\right) = \sum_X P(aX) - \sum_{X \neq Y} P(aX \cap aY) + \sum_{X \neq Y \neq Z} P(aX \cap aY \cap aZ) - \ldots \pm P(\bigcap_X aX) \qquad \text{[A3.8]}$$

The form for the chronic occurrence of stressors is analogous, with *aX* being replaced by *cX*. If the occurrence of stressors is logically independent, then the intersections are replaced by the product of probabilities (Bain and Engelhardt, 1987).

$$P\left(\bigcup_X aX\right) = \sum_X P(aX) - \sum_{X \neq Y} P(aX)P(aY) + \sum_{X \neq Y \neq Z} P(aX)P(aY)P(aZ) - \ldots \pm \prod_X aX$$

It is known *a priori* that the level and duration of the stressor is dependent on the occurrence of the stressor in the first place. Conventionally, the duration of exposure is assumed to be infinity, i.e. a steady state concentration is assumed. The occurrence of acute stress is assumed to be determined by the level of stressor only. In this case, expressed in set theoretical terms the probability of stress is:

$$\text{P}(aX) = \text{P}(aX \cap alx) = \text{P}(aX \mid alx) \,.\, \text{P}(alx). \qquad \text{[A3.10]}$$

Generally though, stressor levels in-stream are variable and consequently the duration of a specific level of stressor is not infinity but of duration τ, where $0 \leq \tau \leq \infty$ or possibly even dynamic. The dynamic case involves mechanistic considerations, which will be considered in Chapter 5. For the purpose of this chapter the level of stressor is assumed to be a function of time but in such a way that τ is long enough for a pseudo steady state to be reached. In analogy to xenobiotics exposure, where it is known that both the level and duration of exposure is important, it is postulated that for all stressors this is true to some extent. Therefore, the expression for the probability if occurrence of stress X due to stressor x should be:

$$P(aX) = P(aX \cap alx \cap adx) = P(aX \mid alx \wedge adx) \,.\, P(alx \cap adx)$$

The level of exposure and duration of exposure are assumed independent. This appears to be reasonable as a first assumption since in general there would be no mechanism that relates the duration and level of exposure. Therefore, the probability of stress becomes:

$$P(aX) = P(aX \mid alx \cap adx) \,.\, P(alx) \,.\, P(adx) \qquad \text{[A3.11]}$$

The problem of determining the risk of unsustainability due to multiple stressors from a single source can be addressed by sequentially solving [A3.10] (or [A3.11] in the case of time-varying concentrations), [A3.9], [A3.6], [A3.7], [A3.4] and [A3.2] (Figure 3.1).

A2.11 NOTES ON THE ESTIMATION OF STRESSOR-RESPONSE RELATIONSHIPS

A2.11.1 TOXIC SUBSTANCE STRESS-RESPONSE RELATIONSHIPS

The aim of this section is to present a method to estimate the parameters for the SRR for toxic substances. In the context used here, toxic substances may refer to any stressor that may be diluted or have its level adjusted when being mixed with water having a different level of stressor. Typically this type of data would be generated by laboratory bio-assessments. Two issues need to be considered: the level of organisation at which the assessment is aimed, the problem of temporally varying stressor levels, and the use of "standard" toxicity benchmarks.

CHOICE OF TEST SPECIES

Not only does the level of organisation within the species of choice matter, but the choice of species also has an influence on the interpretation of derived values. It has become apparent that no single specie can qualify (Kenaga, 1978, Mayer etand Ellersieck, 1986, Blanck *et* al., 1984, Kooijman, 1987). The lowest acute or chronic test result from a set of the most commonly used species, (the alga *Selenastrum capircornutum*, the fish *Poecilia reticulata* and the invertebrate *Daphnia magna*) only managed to come within a factor of 10 of the most sensitive species tested 25% of the time (Sloof *et al.*, 1983; Sloof and Canton, 1983). Therefore, if no single most sensitive species can be found and it is unlikely that a suite of standard test organisms will give an indication of what the susceptibility of the most sensitive species will be like, it could be argued that

a) no species is likely to be significantly more sensitive than the most sensitive test species, or
b) "that differences in sensitivity among species are insignificant unless they are larger than differences among tests of a species-chemical combination" (Suter, 1993), or
c) simply use a safety factor to accommodate all the uncertainty when extrapolating, or
d) assume that species sensitivity will follow some regular distribution and estimate protection levels from that.

The third argument has been in use for some time. The USEPA's uncertainty factor of 10 for taxonomic variance appears to be based on the assumptions that: 1) any invertebrate is a sensitive as *Daphnia* and that any vertebrate is as sensitive as the fish used in the tests, and 2) that protecting a small number of test species 90% of the time is sufficient (Suter, 1993).

The fourth argument recognises the inherent fallacy of the third argument in that there is no evidence that *Daphnia* and fish represent among themselves the most sensitive species or even representative species. The approach used in the derivation of the South African Water Quality Guidelines for the Protection of the Aquatic Environment (Roux, *et al.*, 1996) is based on that used for the calculation of the U.S. National Water Quality Criteria (Stephan *et al.*, 1985) with the exception of the greater emphasis placed on the use of indigenous test species. The approach has been to assume that species sensitivity will follow a regular distribution (in this case a log triangular distribution) and by assuming a level of protection for all species (e.g.95%), a concentration of a toxicant can be calculated. Kooijman (1987) fits a log logistic distribution to toxicity data. However, Suter (1990) considers the choice of distribution to be insignificant in comparison to the more crucial decisions such as level of protection and uncertainties included in the estimation of confidence. It may be argued that all species in a community should be protected and that the selection of any arbitrary protection level does not guarantee protection of ecosystem function. Kooijman (1987) made a similar suggestion. This implies that the criterion value for more and less diverse communities will differ with the more diverse communities having a lower criterion value, since there are more species (and therefore a greater possibility of sensitive species). In contrast, Van Straalen and Denneman (1989) argue that in larger communities the likelihood of functional redundancies is larger and that therefore less restrictive criteria should be applied.

Sensitivity distribution based on species distribution assumes that test species are randomly drawn from the community they are supposed to represent. The argument has been raised that clearly test organisms are not randomly selected, but are usually selected on the basis of ease of laboratory cultivation and happenstance (Cairns and Pratt, 1989). However, ease of laboratory cultivation is determined by species specific knowledge and good laboratory technique rather than by species sensitivity as is borne out by the observation that sensitive species survive and thrive under natural conditions which are considerably more adverse than laboratory conditions. Therefore, unless specifically contraindicated, there would be sufficient reason to assume random selection of species in the toxicity test data to warrant using the data to estimate the probability density function parameters.

Estimating parameters for distributions normally requires a considerable amount of data, which is often lacking. There is considerable need to use extrapolation to derive parameters in sparse data sets. If there are too few data to confidently estimate the parameters of the distribution (such as NOEC, EC_{50} and another percentile < 50) of sensitivity of species for a chemical, it can be estimated by considering the sensitivity data across chemicals where the relevant data are available.

INDIVIDUAL VS. POPULATION BASED ASSESSMENT

The individual based approach in ecology is essentially an application of the reductionist methodology. There are two approaches to follow in conceptualising populations:

1) The **population approach** where the whole population consists of individual organisms that are essentially identical subject to natality and death. An example is the common Lotka-Volterra models used with some success in explaining at a phenomenological level the changes in predator and prey fish caught after the first world war (Braun, 1983 pp 441-449; Suter, 1993). This type of model does not necessarily demonstrate the dynamics involved at a *biologically measurable* level. The parameters in these models (e.g. the predation rate, competition intensity etc.) are mathematical descriptors that are not directly measurable, but can only be inferred or calculated from real population measurements.
2) **Individual based models**, where it is recognised that a population may consist of a number of individuals with different ages, morphological characteristics, fecundity, mortality rates etc. The individual based methods in population ecology explicitly incorporates a knowledge of dynamics and socio-biology of populations in terms of biologically significant parameters such as fecundity, mortality rates or survival probability (Lomnicki, 1992).

A stressor will generally affect different life stages of an organism in different ways, and the effect on the population as a whole can usually not be assessed from "standard" toxicity benchmarks such as the LC50 (Lenski and Service, 1982; Mayer, et al., 1989; Caswell, 1996). The individual-based bio-assessments depend on the testing of a cohort of organisms usually for a relatively small fraction of their natural lifetime. Even chronic toxicity tests do not combine mortality and fecundity data to estimate impacts on a population. In order to do this, though; the life history of the organisms as well as the survival and fecundity rates of a cohort of the organism needs to be known.

A well-established approach to estimate population level effects from individual level observations is by using demographic population models (Caswell and John, 1992). Knowledge of the individual state (i-state) variables such as age size and physiological state are used to derive the population state (p-state). Construction of a population model requires a function that combines the current p-state dynamics and the environment The types of models that could be involved are described in Table A2.1. The discrete-state, discrete-time model described by Caswell (1989) was chosen because the type of data generated in a laboratory bio-assessment appears to fit this model better than the continuous time models.

Table A2.1. Mathematical frameworks for p-state variable models

p-State	Time	Model Type	Reference
Discrete	Discrete	Projection matrices	Caswell, 1989
Discrete	Continuous	Delay-differential equations	Nisbett and Gurney, 1982
Continuous	Continuous	Partial differential equation	Metz and Diekman, 1986

Where individuals can be differentiated on some basis or another, the population projection matrix model Eq. [A2.11] gives the conditional expectation of population number per class (expressed as the vector $\mathbf{n}(t)$):

$$E(\boldsymbol{n}(t+1)|\boldsymbol{n}(t)) = \boldsymbol{A}\cdot\boldsymbol{n}(t) \qquad [A2.11]$$

An inherent advantage in this type of model is the underlying stochastic description of a population already incorporated in the model. From Eq. [A2.11] the assumption of Markov-chain conditions is apparent. This may be a drawback since the future state of a population is not always only dependent on its present state, but may be dependent to some extent on its recent history. As a first approximation the Markov condition may be sufficient. The model can be formulated by a matrix equation Eq. [A2.12].

$$\mathbf{N}_t = \mathbf{A}^t \cdot \mathbf{N}_0 \qquad [A2.12]$$

where: $\mathbf{A} = \begin{bmatrix} F_1 & F_2 & \cdots & F_s \\ P_1 & 0 & \cdots & 0 \\ \vdots & \vdots & \vdots & \vdots \\ 0 & 0 & P_{s-1} & 0 \end{bmatrix}$

and P_i is the probability of survival of members of age class i. Fertility of the population is described in terms of fertility coefficients F_i

A population that responds according to this model will (Caswell, 1989):

1. Eventually reach a **stable age distribution**
2. Grow or decline at a **constant rate**, and
3. Have its long-term behaviour determined by its **dominant eigen value**.

The utility of the transition matrix $\boldsymbol{A}$ in ecotoxicology lies in:

(a) The connection between the dominant eigenvalue of $\boldsymbol{A}$ and the intrinsic rate of population growth. If λ_1 is the dominant eigenvalue of the transition matrix $\boldsymbol{A}$, then $\lambda_1=e^r$ with r the nominal rate of population growth (Caswell, 1989).

(b) The p-state parameters are inferred from easily measured i-state transition variables. In the case of aquatic toxicity tests these are measured in the form of fecundity and survival rates or probabilities.

The SRR parameters can be estimated from an assessment of the population growth characteristics projected from the survival and fertility data collected from individual organisms.

The **upper acceptability limit** (the catastrophic effect level) can be said to be the minimum stressor level corresponding to a zero population growth rate. The rationale for this is that if population numbers are expected to decline in the absence of natural processes such as competition and predation, then the effect could only be expected to be worse in the presence of such factors.

The **lower acceptability limit** (no-observable-effect level) is not as easily assessed since there is no natural cut-off point. In order to generate such a cut-off point it would be necessary to make some value judgements. It could, for example, be argued that any observable decline in population growth rate r would be unacceptable. This r would be the growth rate that could be resolved from the natural population growth rate r_0 with a confidence of, say, 90% (α = 0.1). This rationale is similar to that used in the definition of a toxicity NOEC, subject to the same type criticism, i.e. that statistical significance has nothing to do with ecological significance (Suter, 1993). This argument is valid if there is sufficient ecological knowledge available to estimate an ecologically significant value of r. If not, the statistical value must act as surrogate for ecological significance.

Eq [A2.12] represents a general population growth assessment. In order to use this type of model, there are two types of parameters that need to be calculated or estimated: the age-specific **probability of survival** and the age-specific **fertility functions**.

Survival probability estimate

One of the most powerful means to generate these data is by using hazard analysis (Cox and Oakes, 1984). A hazard model relates the probability of a transition occurring (as the dependent variable) to a causal factor (as the independent variable). If $f(t)$ is the instantaneous probability of an event occurring at time t and $F(t)$ is the cumulative probability of the event having occurred before time t, then the hazard function, $\mu(t)$, for example the probability of an organism dying in between t and $t+dt$ is given by (Caswell, 1989), is given by Eq. [A2.13].

$$\mu(t)dt \approx -dt\frac{\partial \ln l(t)}{\partial t} \qquad \text{[A2.13]}$$

where $l(t)$ is the probability of surviving to time t. Generally, the probability of surviving to time t give exposure to concentration x, $S(t/x)$, is related to the hazard function $h(t/x)$ by (Namboodiri and Suchindaran, 1987; Moore, *et al.*, 1990):

$S(t/x) = \exp[-\int_0^t h(t/x)\partial t]$. The hazard function $h(t|x)$ is also called the force of mortality and is equivalent to $\mu(t)$ used by Caswell (*op cit.*). From Eq.[A2.13] the probability of survival over the interval $t+\Delta t$ is given by Eq. [A2.14].

$$\frac{l(t+\Delta t)}{l(t)} = e^{-\mu(t)\Delta t} \qquad \text{[A2.14]}$$

Using a proportional hazards model, the fraction (probability) survival under a given exposure regime $S_1(t)$ can be related to the baseline survival $S_0(t)$ by (Namboodiri and Suchindaran, 1987): $S_1(t) = S_0(t)^{\exp[f(\beta)]}$.

In order to parameterise the population transition matrix $\boldsymbol{A}$ of Eq. [A2.12] it is necessary to estimate $S_1(t_i)$ for each time interval t_i and each life stage modelled in this matrix. There are two options to estimate the survival:

a) by **direct calculation** from suitable **experimental data** (e.g. from toxicity bio-assessment) where $S_1(t_i)$ and $S_0(t_i)$ can be calculated from the exposed and control runs respectively, or
b) by **indirect estimation** when no suitable life table experimental data are available where $S_1(t_i)$ must be calculated from **other ecotoxicological data**.

Direct calculation from bio-assessment data

By curve fitting the parameters for the proportional hazards model could be determined. Moore, *et al.* (1990) tested a model of the form $P_i(x) = P_{0,i}^{\exp[\beta_i(x-x_0)]}$ and showed that for three tested pesticides the potency β remained constant through all intervals, and hence β_i can be replaced by

β. Here $S_i(x)$ is the probability that a test animal alive at the beginning of the i^{th} interval will survive to the end of the i^{th} interval, β_i is the potency of concentration x during the i^{th} interval, x_0 is an arbitrarily chosen log concentration to centre the observations and P_i is the underlying conditional probability of survival at the centring concentration x_0.

$$S_i(x) = \left(\prod_{j=1}^{i} P_j(x) \right)^{\exp[\beta(x-x_0)]} \qquad \text{[A2.15]}$$

Bio-assessment data that would be applicable for this kind of estimate would result from experiments where a suitable life table can be generated. This would mean that:

- the exposure would encompass practically the whole life cycle of the organism, or at least that part of the life cycle spent in water, and
- both mortality and fertility data need to be recorded, which means that range of exposure levels need to be wide enough.

Indirect estimation from other ecotoxicological data

The survival can be estimated from fundamental ecotoxicological data such as the uptake and excretion rates, the lethal body burden and the log K_{ow} of the substances involved. The methodology is similar to survival time analysis. The toxicokinetics become important when estimating the fraction of a population surviving to a given time. The time would typically correspond to the cohort age structure used to discretise the lifetime of the organism. The calculation uses the same type of data used to estimate the effect of temporally varying concentrations.

THE PROBLEM OF TEMPORALLY VARYING COMPOSITION

In the derivation of substance specific criteria bio-assessment data was used that selected the standard test durations (e.g. 48 hours for many of the smaller invertebrates and 96 hours for larger animals). In these tests the levels of substances were kept constant. Stressor levels cannot be expected to be constant in real situations. This begs the question of what happens when stressor levels vary. The approach in the application of the USA criteria has been to use 1-hour average concentrations when considering acute substance specific criteria and to use 4-day average concentrations when using chronic criteria (Delos, 1994).

In order to clarify the role of time in the effect assessment of substances, the toxicokinetics need to be considered. This involves determining the mode of action (MOA). Depending on the classification used anything between two and eight MOA's can be distinguished (Verhaar, *et al.*, 1999). These may include the narcotics, polar narcotics, electrophiles and reactive or receptor mediated compounds. Among non-metal toxicants the polar narcotics probably represent the most rapidly excreted substances and the reactive chemicals the least excreted compounds. Mechanistically these classes are distinct and a comparison appears in Table A2.2

Table A2.2 Comparison of polar narcosis and reactive toxicity (Legierse et al.*, 1999; Freidig,* et al.*, 1999; Verhaar,* et al.*, 1999)*

Aspect	**Polar Narcosis**	**Reactive toxicity**
Receptor interaction:	Reversible	Irreversible
Toxicodynamics determined by:	Cell membrane	Intracellular chemical pool
Dose metric	Internal concentration	Area under concentration vs. time curve (AUC)
Critical physiological parameter	Critical body residue (CBR) or lethal body burden (LBB) = constant for all chemicals in class	Critical area under curve (CAUC) = constant (CBR is temporally variable)
$EC_{50}(t)$ determined by:	Bioconcentration kinetics	Cumulative inhibition of receptor
Model	CBR	Critical Target Occupation (CTO)
$LC_{50}(t)$ =	$\frac{LBB}{BCF \cdot (1-e^{-k_2 t})} = \frac{LC_{50}(\infty)}{(1-e^{-k_2 t})}$	$\frac{CUAC_a}{t} + LC_{50}(\infty)$
LBB =	$LC_{50}(\infty) \cdot BCF$	$BCF \cdot (1-e^{-k_2 t}) \cdot LC_{50}(t)$

With complex effluents, variables such as the LBB cannot be determined unless the effluent composition is known; an exercise that would partially defeat the purpose of using WET assessment in the first place. However, from the expressions in Table A2.2, there is a relationship between the LBB and the $LC_{50}(\infty)$. For the purpose of evaluating the age-specific cumulative fractional mortality it is necessary to know $LC_S(t)$.

Mancini (1983) developed a simple toxicokinetic approach to estimate effect for time varying concentrations. Based on the assumptions that:

1. Variation in survival times defines a distribution of sensitivity
2. At any concentration the same percentile survival time defines a common sensitivity level, and
3. All organisms with similar sensitivity have similar regulatory characteristics

and using a simple single compartment model for the target organism:

$$\frac{dy(t)}{dt} = k_1 \cdot x - k_2 \cdot y(t) \qquad \text{[A2.16]}$$

where y = intra-organism concentration of the toxicant [mass toxicant/mass organism]

x = concentration of toxicant in the water [mass toxicant/volume water]

k_1 = uptake rate [volume water/(mass organism*time)]

k_2 = depuration rate [/time]

with the boundary values:

$y(0) = 0$

$y(t') = d$ (lethal dose)

where: t′ = time to death. If at first it is assumed that the concentration is constant for a period it was shown that:

$$y(t) = \frac{k_1}{k_2} \cdot x \cdot \left[1 - e^{-k_2 t}\right] \qquad \text{[A2.17]}$$

Recognising that $BCF = \frac{k_1}{k_2}$ Eq [A2.17] rearranges to Eq. [2.18].

$$y(t) = BCF \cdot x \cdot \left[1 - e^{-k_2 t}\right] \qquad \text{[A2.18]}$$

When the intrabody dose, $y(t)$, reaches a level referred to as the critical body residue (CBR) or lethal body burden (LBB), the organism dies. The implication is that different chemicals with a narcotic mode of action will display an additive body burden, which, on reaching the LBB for the organism, will result in death of the organism. (Sijm *et al.*, 1993). For anaesthetic chemicals the LBB appears to vary between 2 and 8 mmol/kg irrespective of structure.

SOME EXPRESSIONS FOR THE BODY RESIDUE OF NARCOTIC SUBSTANCES UNDER TEMPORALLY VARIABLE WATER CONCENTRATIONS

For **pulsed toxicant concentration with a square waveform**: with water concentration x for $0<t<t_0$
and x=0 for $t_0<t<t_1$. Then:

$$C(t_1) = \frac{y(t_1)}{u} = \frac{x}{r} \cdot \left[e^{-r(t_1-t_0)} - e^{-rt_1}\right] \qquad \text{[A2.19a]}$$

or (Mancini, 1983)

$$C(t_1) = \frac{y(t_1)}{u} = \frac{x}{r} \cdot \left[1 - e^{-rt_1}\right] + C(t_0) \cdot e^{-rt_1} \qquad \text{[A2.19b]}$$

This corresponds to a situation where depuration takes place when the external concentration drops after uptake of toxicant at the higher ambient toxicant concentration (Figure A2.1). If toxicant build-up takes place long enough, then that fraction of organisms for which the equivalent dose, $C(t)$, equals or exceeds the equivalent mortality dose, D, die.

This could have a significant effect on the mortality of the organism. Considering Figure A2.1, if the equivalent mortality dose is 10 mmol/kg, then the expected survival time for the 10 percentile of organisms is about 14 days, the median survival time is about16 days, but the 90th percentile of organisms in this exposure scenario is ∞.

More generally, if the aqueous concentration varies in a stepwise manner with changes at discrete time points t_i a fixed time interval t_d apart and with the concentration remaining constant during this period at x_t, then the internal concentration at the end of the interval is given by (Kooijmans, 1994):

$$y_{t+1} = e^{-rt_d} \cdot y_t + (1 - e^{-rt_d}) \cdot \frac{x_t \cdot u}{r} \qquad \text{[A2.20]}$$

If x_t follows a random increment process, then solution of the stochastic analogue of the differential equation [5.10] yields the expected value of $y(t+1)$ is:

$$E[y_{t+1}] = (e^{-rt_d})^{t+1} \cdot E[y(0)] + (1 - e^{-rt_d}) \cdot \frac{u}{r} \cdot E[x_t] \cdot \sum_{j=0}^{t} (e^{-rt_d})^j \qquad \text{[A2.21]}$$

and

$$\mathrm{var}[y_t] = \mathrm{var}[x_t] \cdot \left(\frac{u}{r}\right)^2 \cdot \frac{1 - e^{-rt_d}}{1 + e^{-rt_d}} \qquad \text{[A2.22]}$$

In continuous time the expected value of $y(t)$, $E[y(t)]$, is the same as $E[y_t]$ in equation [A2.21] and:

$$\mathrm{var}[y(t)] = \mathrm{var}[x_t] \left(\frac{u}{r}\right)^2 \left(1 - \frac{1 - e^{-rt_d}}{rt_d}\right) \qquad \text{[A2.23]}$$

For water **concentrations with an exponential decay** function (peak concentration A and decay constant k), i.e. $x(t) = A.e^{-kt}$, :

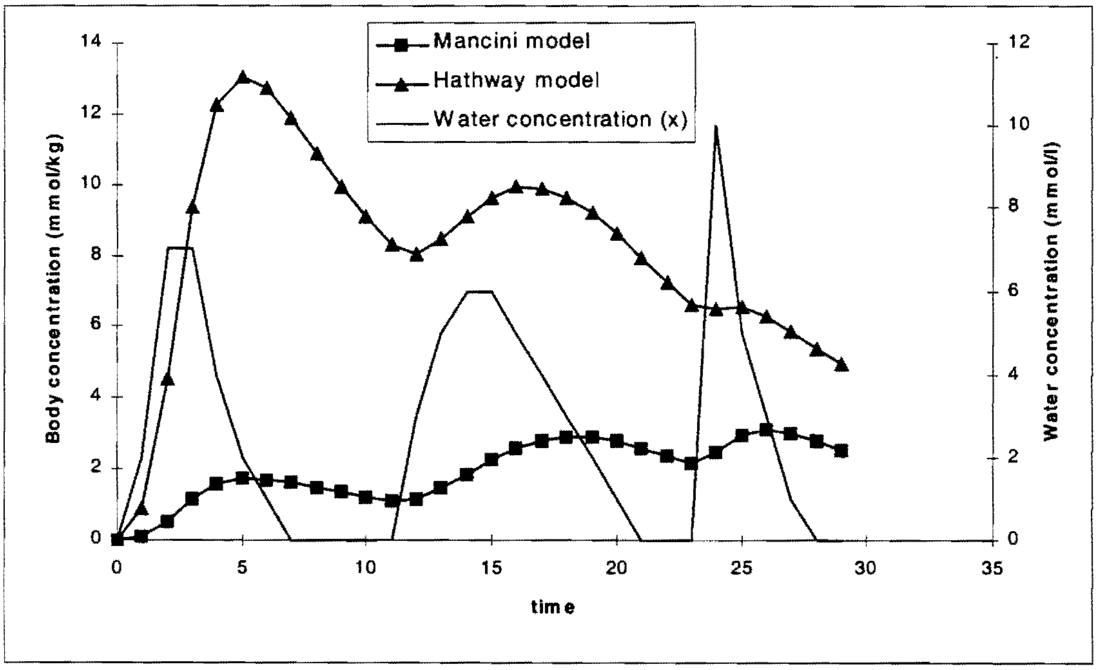


Figure A2.1 An illustration of the importance of knowledge of mechanisms of toxicology. The body burden of a hypothetical substance with k2=0.09 and BCF = 1.11 predicted by the Mancini and Hathway models (Eqs. [A2.19a] and [A2.26] respectively) as a function of the substance concentration in water. The Mancini model predicts a more rapid response to changes in aqueous concentration.

$$D = \frac{d}{u} = \frac{A}{(r-k)} \cdot \left[e^{-kt'} - e^{-rt'}\right] \qquad \text{[A2.24]}$$

and

$$C = \frac{y(t)}{u} = \frac{A}{(r-k)} \cdot \left[e^{-kt} - e^{-rt}\right] \qquad \text{[A2.25]}$$

The uptake of a substance has so far been assumed to be instantaneous. This would generally not be true and Hathway (1984) suggested that the equilibrium concentration may be described by:

$$\frac{dy}{dt} = x \cdot u \cdot e^{-ut} - ry \qquad \text{[A2.26]}$$

The effect of this model is that the organism does not immediately respond to a change in concentration. If the dosed concentration, x, is a function of time, x(t), then a lagging of intra-organismal concentration of the toxic substance can be expected. This is demonstrated in Figure A2.2.

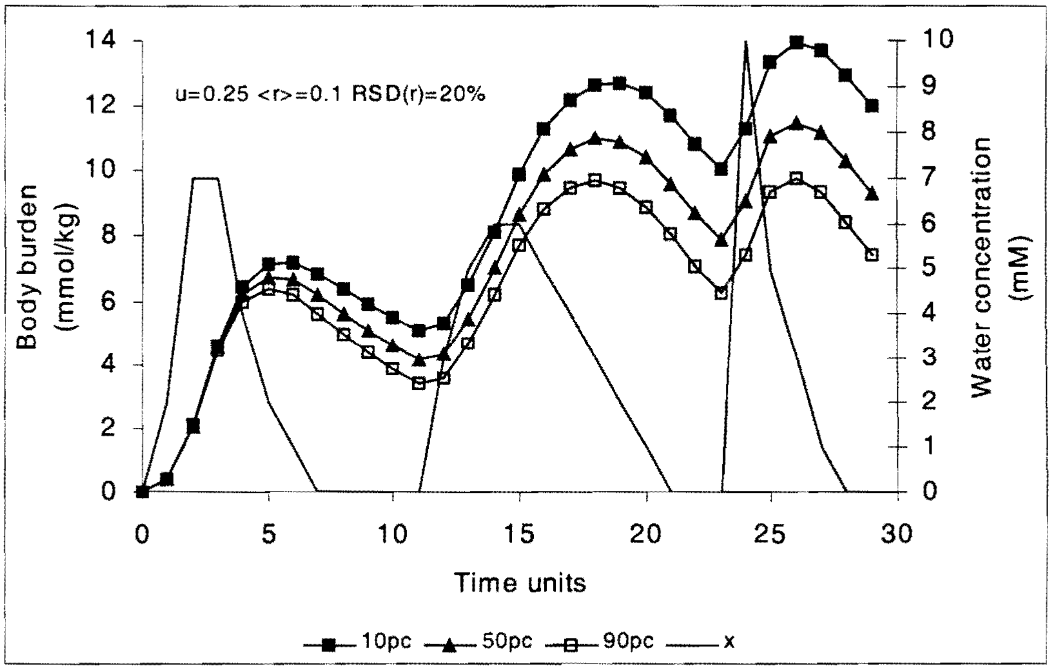


Figure A2.2 Demonstrating the effect of variability in individual organism depuration rate on the expected body burden within for a population as a function of concentration in water. Body residue of a hypothetical substance in an organism with an average k2 = 0.1 and standard deviation = 0.02. The average BCF = 4.

A further refinement can be attained by recognizing that the substance(s) absorbed may not in themselves be toxic and that further reaction inside the organism, whether by activation or binding to a target receptor, may be required to see an effect. For a reaction between dosed substance A and intra-organismal substance B:

$$A + B \underset{k_3}{\overset{k_2}{\rightleftarrows}} AB \qquad \text{[A2.27a]}$$

Then, with respect to A, at a nominal concentration y_1, the concentration of the reaction product (AB) y_2 is given by:

$$\frac{dy_2}{dt} = k_2(y_1 - y_2) - k_3 y_2 \qquad \text{[A2.27b]}$$

If the kinetics is determined by the concentration of y_1, i.e. the uptake of the toxicant is rate determining, then the effect will be determined by Eq. [A2.27c]. If the concentration of the receptor, b, is rate determining then the effect will be determined by [A2.27d]

$$y_2 = \frac{k_2 y_1}{k_2 + k_3} \cdot \left[1 - e^{-(k_2+k_3)t}\right] \qquad \text{[A2.27c]}$$

$$y_2 = \frac{k_2 b}{k_2 + k_3} \cdot \left[1 - e^{-(k_2+k_3)t}\right] \qquad \text{[A2.27d]}$$

The dynamics of toxic effect of substance A applied *in aquo* at concentration x is give by the system of equations [A2.28].

$$\begin{aligned} \frac{dy_1}{dt} &= u \cdot f \circ x(t) - r \cdot y_1(t) \\ \frac{dy_2}{dt} &= k_2(y_1 - y_2) - k_3 y_2 \end{aligned} \qquad \text{[A2.28]}$$

with the driving function $f(x)$ taking on a suitable form.

Alternative to Mancini's assumption that there is a distribution of regulatory efficiency that gives rise to variability in response, it could be argued that regulatory efficiency is constant but that there is distribution of receptor site density over a population, i.e.. that b in Eq. [A2.27d] is stochastic variable.

FURTHER RESEARCH

In the case of single substances, the above approach is simple to quantify in principle since the body burden of an identifiable substance can be measured and k_2 and BCF can be calculated. The problem arises in predicting the effect of temporally varying complex effluents. As shown in the foregoing illustration the body burden of a substance in an organism varies with varying ambient concentration.

The problem that needs to be solved is how to estimate the body burden of lethal components of a complex mixture from toxicity bio-assessments. If it is assumed that the components of a mixture interacts by the narcotic mechanism, then at the time of death of an organism, the narcotic substances that had partitioned from the mixture and of which the organism cannot excrete fast enough, will total to the LBB. It seems reasonable to suppose that a complex effluent will have an apparent k_2 value. If this value is known, then Eq. [A2.25] (or A2.19 to A2.26 above depending on the situation) can be used to estimate the apparent body burden of the mixture (effluent). If it is recalled that k_2 is a stochastic variable for a population, then the probability distribution of mortality can be estimated and from that the organisms population growth can be estimates (subject to assumptions or measurements about it fertility). The two critical questions that need to be answered are:

- How can the apparent BCF of a complex mixture be estimated?

- Can the differential excretion rate for the components be estimated from measurement other than by temporally variable toxicity-bioassessment?

In both cases the development work on biomimetic extractions seems encouraging (Verbruggen, *et al.*, 1999) and could be investigated further.

A2.11.2 HABITAT- AND FLOW-STRESSOR-RESPONSE RELATIONSHIPS

The prediction of biological effect is notoriously difficult and yet the need for prediction is very real (Armitage, 1994). The problem of flow and habitat stress assessment has been presented in Chapter 3 as a strong reason for the use of fuzzy set theory. The reason being that often there is no controlled experimental evidence to derive the SRR parameters. These parameters are estimated based on the assessment of an expert based on analogy, limited observation etc. The situation is analogous to what is described by Klir and Folger (1988) as an interpersonal communication problem. The stressor risk assessment can be formulated in the form $E = R \circ A$ where R is the fuzzy relationship between fuzzy stressor situation analysis A and the fuzzy expectation of effect E and o is a suitable implication operator. In the examples presented in this study, R has been simplified to crisp relationship but this need not generally be so.

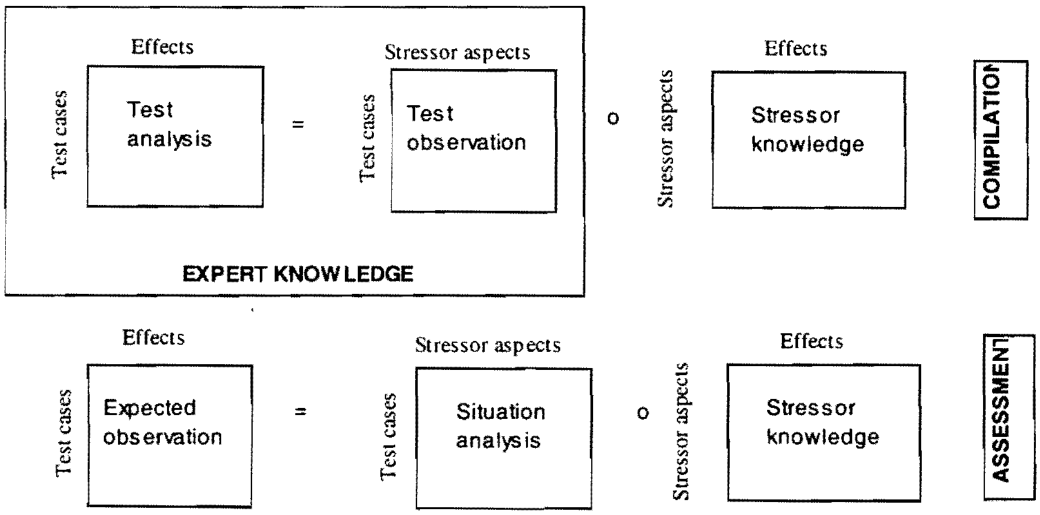


Figure A2.3. Schematic of the possible use of fuzzy sets in assessing fuzzy expectation. Stressor-effect relationships are encapsulated in the stressor knowledge block. The upper schematic follows a logic from left to right (i.e. the stressor knowledge is generated) while in the lower schematic the goal is deriving the expectation of effect for a give number of cases.

The expectation assessment problem resolves into two practical problems: 1) Deriving the relationship R expressing the knowledge of response of stressors, and 2) incorporating new ob servational evidence to update the expectation E.

NOTES ON THE FUZZY RELATIONSHIP R

R is derived from a training set of stimuli and responses collected over as wide a range as possible of test cases. The process is described in Figure A2.3

Both R and *X* are derived from and informed by the interpretation of real data by the ecologist/ ecotoxicologist. Consider the situation where the stressor is characterised by characteristic set $X = \{x_1, x_2, \ldots x_n\}$ while the effects are characterised by set $Y = \{y_1, y_2, \ldots y_m\}$. The experience of the ecologist in dealing with a particular stressor derives from observations in a number of test cases with corresponding stressor situation analyses. For every stressor metric $x \in X$ there is an observed or inferred response $y \in Y$ in the set of test situations $T = \{t_1, t_2, \ldots t_k\}$. Each test case t results in a stressor knowledge matrix:

$$R_t = \begin{bmatrix} \mu_R(x_1,y_1) & \cdots & \mu_R(x_1,y_m) \\ \vdots & \mu_R(x_i,y_j) & \vdots \\ \mu_R(x_n,y_1) & \cdots & \mu_R(x_n,y_m) \end{bmatrix}$$

The elements of the knowledge matrix R can be evaluated in two different ways resulting in two different knowledge bases:

- An occurrence relation R_o that corresponds to the answer to the question: "How often does stressor characteristic *x* occur in conjunction with effect *y*?" This is derived from an assessment over all the test cases of the frequency of the co-occurrence of *x* and *y*, or,
- An confirmatory relation R_c that corresponds to the answer to the question: "How strongly does effect *y* confirm the presence of stressor characteristic *x*?". This results from an analysis of the correlation of the intensity of *x* and the intensity of *y*.

This approach could be considerably expanded, both in terms of the information content of the knowledge base and the modelling of, and expert query to construct the relationship (see e.g. Yager (1992)).

EXAMPLE: FLOW-RELATED EFFECT ASSESSMENT

Consider a flow-related stressor characteristic set $=\{$*sufficient water depth (d), correct flow timing (t), adequate scour flow(s)*$\}$ and the effect characteristic set $=\{$*adequate fish community maintenance (f), adequate invertebrate community maintenance (c), physical stream habitat maintenance (h), refugia maintenance (r)*$\}$. The linguistic qualifiers and their membership interpretation are listed in Table A2.2.

If the modifier "very" needs to be added to the qualifier then the modified membership function, $\mu_A'(x) = 1.5*\mu_A(x)-0.25$. Assume a knowledge base in form :

d and t are very often important for f ,

f is always important for c,

s is seldom important for f,

s is often important for c,

s is always important for h,

d is sometimes important for h,

t is seldom important for c,

t is never important for h, etc.

Table A2.2 Linguistic qualifiers and their membership grade evaluation

Characteristic (x or y) qualifier	$\mu_A(x)$
Never	0
Seldom	0.25
Sometimes	0.5
Often	0.75
Always	1

From these data a relationship can be constructed:

$$R = \begin{array}{cccccc} & f & c & h & r \\ d & 0.75 & 0.75 & 0.5 & 0.25 \\ t & 0.875 & 0.25 & 0.75 & 0.125 \\ s & 0.125 & 0.875 & 1 & 0.125 \end{array}$$

When a specific flow scenario is being assessed, the probability distribution for the flow characteristics might be assessed from the knowledge of the catchment size and topography, rainfall record or from actual measurements. The values of $\mu_A(x)$ will likely be derived from an expert assessment of when the measured or predicted flow corresponds to sufficient depth, suitable timing and adequate scour flow. A typical example of this type of expert knowledge encapsulation might be as shown in Table A2.3. This implies that a relationship exists that expresses $\mu_A(x)$ as a function of flow. A typical flow assessment A might be:

$$A = \frac{d \quad t \quad s}{0.5 \quad 0.7 \quad 0.2} \text{ if } d = 30 \text{ cm}, t = 7.6 \text{ weeks and } s = 6.8 \text{ m}^3\text{s}^{-1}.$$

If the max-min composition is used as implication operator then the expected effect will be:

$$E = \frac{f \quad c \quad h \quad r}{0.7 \quad 0.5 \quad 0.7 \quad 0.25},$$ which means that refugia maintenance is most likely to be affected by the affected flow scenario. If all effects are assumed to be equally important in determining the end-point effect (e.g. loss of sustainability), the possibility for the end-point will be $(1\text{-min}(\mu_A(x)) = 0.75$.

Table A2.3. Example of a possible format of membership functions for flow stressor characteristics.

Characteristic	Metric	Function
Depth (d)	Average flow depth (cm)	$\mu_A(d)=\begin{cases}1 & if\ d>50\\ \frac{50-d}{40} & if\ 10\le d\le 50\\ 0 & if\ d<10\end{cases}$
Timing (t)	Displacement of expected peak flow (weeks)	$\mu_A(t)=\begin{cases}1 & if\ t<2\\ \frac{t-2}{8} & if\ 2\le t\le 10\\ 0 & if\ t>10\end{cases}$
Scour flow (s)	Minimum flow rate ($m^3.s^{-1}$)	$\mu_A(s)=\begin{cases}1 & if\ d>0.8\\ \frac{0.8-s}{0.6} & if\ 0.2\le s\le 0.8\\ 0 & if\ s<0.8\end{cases}$

EXAMPLE: ESTIMATING ACCEPTABLE STRESSOR VALUES.

The same data as in the previous example applies. In order to derive management criteria, the process for the assessment above is reversed in that an acceptable level of effect is specified while the corresponding stressor level is required. Say that a level α of effect is considered acceptable. That means that $\mu_E(y) = \alpha$, which implies that $\alpha = \max_{x\in X}[\min(\mu_A(x), \mu_R(x,y))]$.

This means that $\min_{x\in X, y\in Y}\{\mu_A(x), \mu_R(x,y)\} \le \alpha$ or,

$$\mu_A(x) \le \begin{cases}\alpha & if\ \mu_R(x,y)\ge\alpha\\ \mu_R(x,y) & if\ \mu_R(x,y)<\alpha\end{cases} \qquad [A2.29]$$

Therefore, if $\alpha = 0.2$ then $\mu_A(d) \le 0.2$, $\mu_A(t) \le 0.125$ and $\mu_A(s) \le 0.125$, which translates to d = 42 cm, t = 3 weeks and s = 0.725 $m^3.s^{-1}$.

A11.3 INTEGRATING BIOMONITORING IN ECOLOGICAL EFFECT EXPECTATION

The previous section had shown that the estimate of $\mu_A(x)$ is very important in both effect assessment and stressor value assessment. The function parameters illustrated in Table A2.3 will determine to large extent what the outcome a calculation will be. At the outset, before any site-specific data are available, these parameter values stem from analogy or even educated guessing. In either case there is room for uncertainty in the parameters.

For flow-related or habitat stress, it is unlikely that experimental values will (generally) be available. However, a number of biotic indices have been developed that pronounce on the stressor impacts to greater or lesser extent (Metcalfe-Smith, 1994, Kleynhans, 1999a). These data are often the only indication of *in situ* effect that is available for estimating SRR's. These biomonitoring data may be therefore be useful in informing and updating effect.

This situation may be modelled as being analogous to the combination of evidence from evidence theory. An application of Dempster's rule of combination (Eq. [A2.30]) as described in Klir and Folger (1988) will be used to illustrate how biomonitoring results can be used to update SRR parameters (see also Smets, 1991a, b and c).

$$\mu_{12}(A) = \frac{\sum_{B \cap C = A} \mu_1(B) \cdot \mu_2(C)}{1 - \sum_{B \cap C = \varnothing} \mu_1(B) \cdot \mu_2(C)} \qquad [A2.30]$$

where two independent sets of evidence (or expert opinion) on sets A, B and C.

Consider the case where there are fish community integrity (*fi*) data and invertebrate community integrity (*ii*) available and instream habitat integrity (*hi*) data. These data may be interpreted by an expert as indicating that the SRR must be adjusted (set D) to indicate lower effect (L), higher effect (H) or no substantial change (N). The combined evidence can be used to generate a membership function for each set as indicated in Table A2.4 below.

Table A2.4 Evaluating the membership from biomonitoring data.

Biomonitoring qualitative indication	Change assessment	Membership
↑↑↑ or ↓↓↓	Definitely	1
↑↑ - or ↓↓ -	Likely	0.75
↑ - - or ↓ - -	Maybe	0.5
↑↓ -	Unlikely	0.25
- - -	No	0

↑, ↓ and – indicate evidence upward, downward and no adjustment respectively.

If the modifier "very" needs to be added to the qualifier then the modified membership function, $\mu_A'(x) = 1.5 * \mu_A(x) - 0.25$. For the purpose of this evaluation it is assumed that L ∪ H (lower or higher) and L ∪ N ∪ H (lower or higher on no change) are empty sets.

It is now assumed that the current parameter set is the accepted set since no *a priori* evidence exists that this set should be changed in any particular way. This is interpreted to mean that the evidence is equally distributed over all the changes that need to be made and therefore $m_1(D)$ =

0.2 (i.e. the evidence is equally distributed over the 5 cases in Table A2.4). The other evidence for change ($m_2(D)$) is derived from the biomonitoring data membership $\mu_D(x)$ (Table A2.4). In order to meet the requirement for evidence that $\sum m(X) = 1$, $\quad m(X) = \dfrac{\mu_D(x)}{\sum \mu_D(x)}$

An example of an update is provided in Table A2.5.

Table A2.5 An example of evaluating evidence for the change of SRR parameters.

Change	m_1	μ_D	m_2	m_{12}
L	0.2	0.75	0.4	0.45
H	0.2	0.125	0.07	0.08
N	0.2	0.5	0.27	0.18
L ∪ N	0.2	0.25	0.13	0.28
H ∪ N	0.2	0.25	0.13	0.01

The implication of the values in Table 2.5 is that SRR parameters are most likely to be adjusted for lower response but they might also stay the same. As a first (unsophisticated) approach parameter values in Table A2.3 might be iteratively adjusted until m_{12} in table A2.5 indicates neutrality with respect to the need for adjustment.

The indications are that the Dempster-Schafer approach can be used to update the SRR's of flow and habitat related stressors from biomonitoring results. The details of these procedures need to investigated.

APPENDIX TO CHAPTER 4

NOTES ON THE SOLUTION TO THE DSMS PROBLEM

A4.1 CODING OPTIONS IN THE SOLUTION OF THE DSMS PROBLEM BY GENETIC ALGORITHM OPTIMISATION

The formulation of the optimisation problem is described in Paper 4. The problem was coded in MS-DOS QBasic (Version 1.1). This choice of coding language was solely dictated by familiarity and not by any considerations of efficiency of programming. The coding for the various versions of the algorithms is listed in the Addendum.

Four versions of the genetic algorithm coding were produced. The approaches and their differences are described in Table A4.1.

Table A4.1 Differences in versions of the genetic algorithm for the solution of the catchment optimisation problem investigated in this study. Coding name refers to listing in the Appendix of this chapter.

Coding name	Attenuation satisfaction (λ_x)	Equity constraint used?	Control parameter initialisation distribution
G1A	Average {source minima}	No	Uniform from focussed or shifted parameter domain
G1B	Average {source minima}	No	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.

G2B	Average {source minima}	Yes	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.
G3B	Inf{source minima}	Yes	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.
G4B	Inf{source minima}	No	Exponential distribution EXP(λ) such that $\lambda = \ln(0.5)/\mu$ where μ is the centre of the focussed or shifted sampling domain.

A4.2 RESULTS

APPLICATION OF A GENETIC ALGORITHM TO THE CATCHMENT DSMS PROBLEM

The results of algorithm convergence and the control variables corresponding to the best λ value are shown in Figures A4.1 to A4.7.

Comparison of Figures A4.1 a) and b) indicates that there are probably two minima with λ values 0.54 and 0.74 with the latter probably representing the optimum. There is a slight improvement in the rate of convergence of the algorithm using all exponential distributions to assign initialising values to control variables. The probability of finding the optimum is slightly lower in the former. Comparison of the optimal attenuation values indicates similar performance. The slightly better convergence rate favoured using the exponential distribution in further work. Comparison of λ with λ_X and λ_R (not showed here) indicated that λ_X was the dominant factor in determining λ.

The argument might be made that optimisation with the constraints as given treats different sources of the same stressor differently. Including the equity constraint produced results as shown in Figure A4.2. The addition of the equity constraint significantly reduced the rate of convergence (Figure A4.2 c) and the attenuation values bears little resemblance to the basic algorithm results (Figure A4.2 b) and Figure A4.2 d), but the tendency for same stressors to converge to similar values is apparent. The best λ decreased from 0.74 to 0.15. Analysis of λ contributions indicated that λ_X still dominated λ.

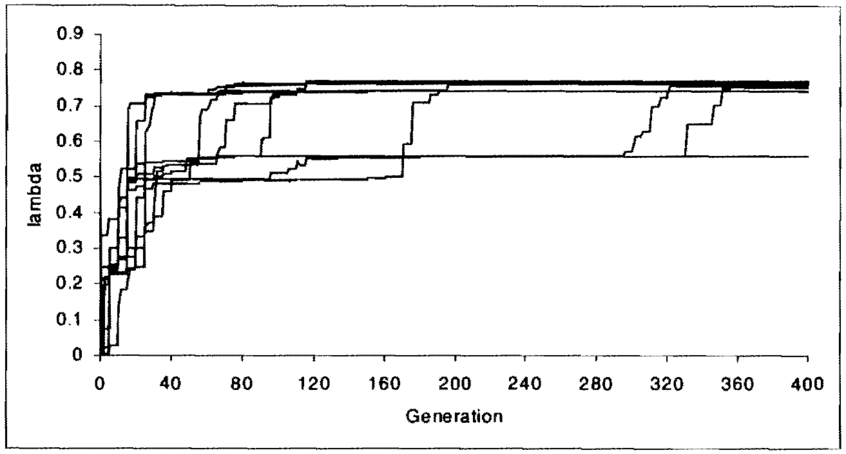
The problem might still arise that if the arithmetic average minimum λ_X is used as an aggregation measure, that some sources may have 0 acceptability while other have a high acceptability.

Addition of an overall minimum acceptability as criterion for λ (i.e. that corresponds to a conjunction of all source and stressor λ_x values) produces the results depicted in Figure A4.5. This shows that the best λ is still lower (about 0.1) and anomalous behaviour of the flow-stressor attenuation.

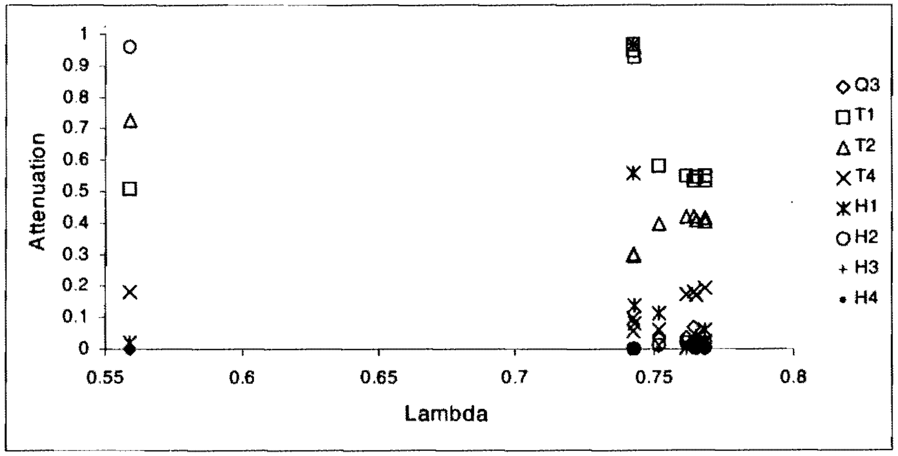
The apparently obvious next step, combining the average minimum λ_x aggregation without an equity constraint produced degenerate $\lambda = 0.99$ for all runs in all scenarios within no more than 80 generations. Figure A4.9 a) to g) shows the variability in the best stressor attenuation values indicating no tendency for stressor-source specific attenuation to converge (the exception being xH4, which was consistently zero).

Figure A4.6 compares the scenario where the toxic attenuation acceptability range was reduced. The attenuation values in comparison to the baseline showed the inherent danger of using average minimum aggregation. The overall λ only decreased very slightly. When using the conjunction aggregation, λ decreased to about 0.09 but when using the conjunction aggregation with equity constraints the stressor specific attenuation remained essentially the same with toxics attenuation being slightly lower. This might be an artefact of the membership function, which asymptotically approaches 0 and 1.

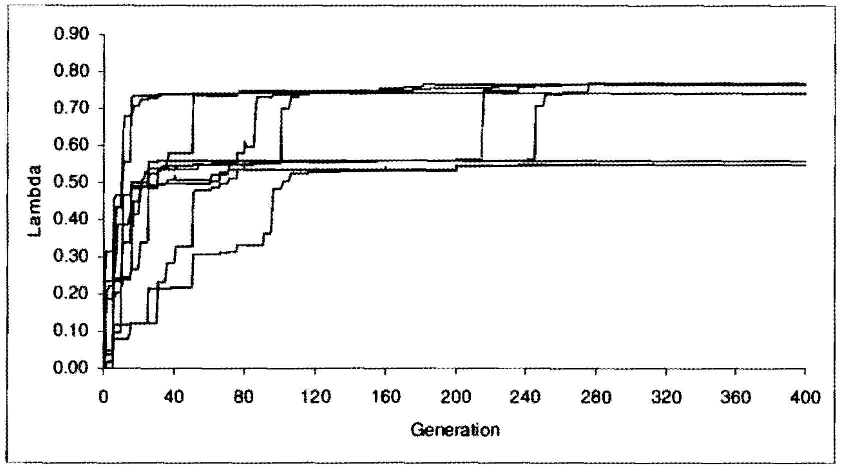
The impact of placing lower risk constraints on the optimal solution resulted in the data depicted in Figure 4.7. When no equity constraints were used and in the absence of conjunctive aggregation, source 1 is heavily penalised. When both types of constraints are added (Figure A4.8), λ comes down to about 0.01 with λ_x still being dominant with λ_{eq} closely following. Interestingly enough, the risk constraint (in terms of λ_R) has very little direct impact on λ.



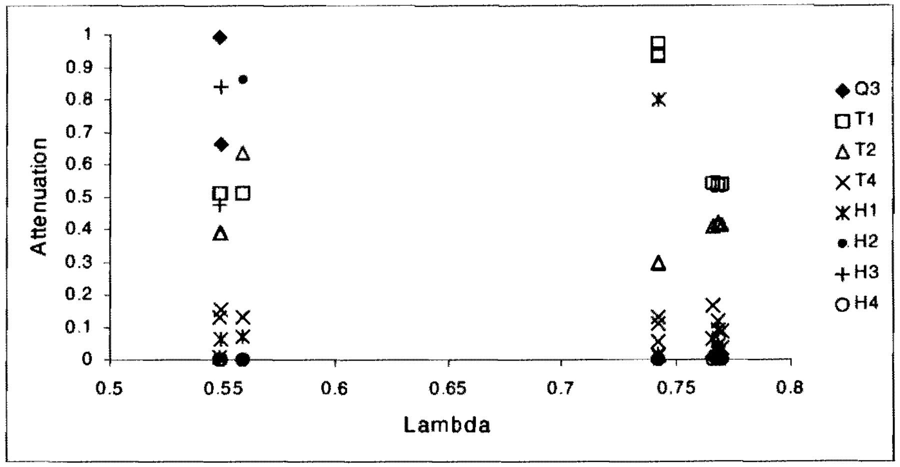
a) λ generated by Code G1A



b) Stressor attenuation as a function of λ generated by Code G1A

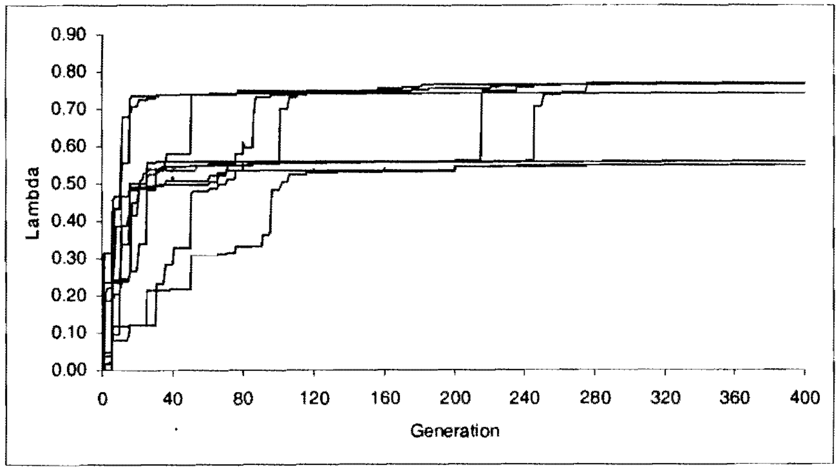


c) λ generated by code G1B

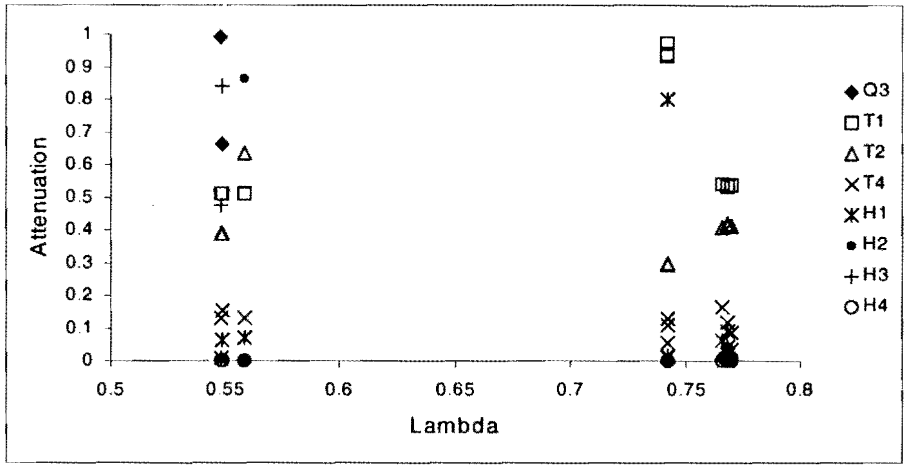


d) Stressor attenuation as a function of λ generated by Code G1B

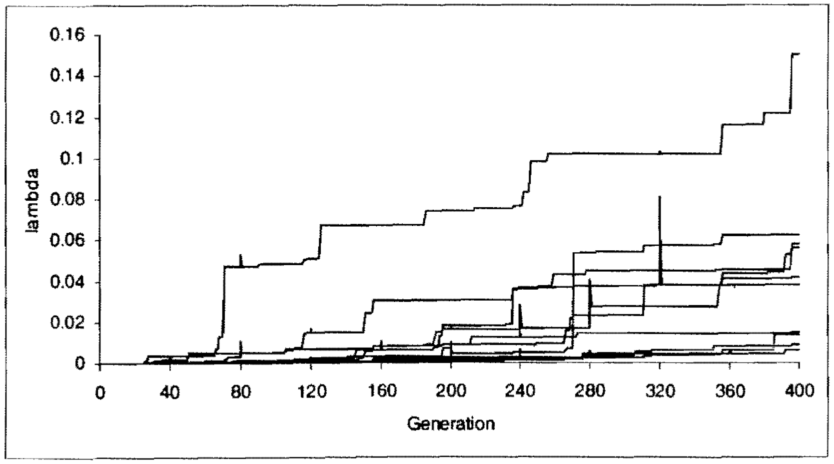
Figure A4.1 A comparison of the effect of control variable initialisation distribution on the performance of the genetic algorithm.



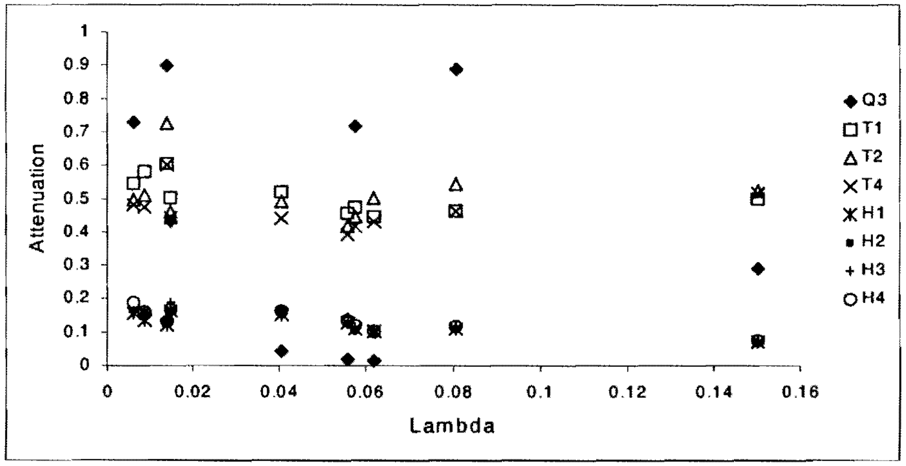
a) λ generated by Code G1B



b) Stressor attenuation as a function of λ generated by Code G1B

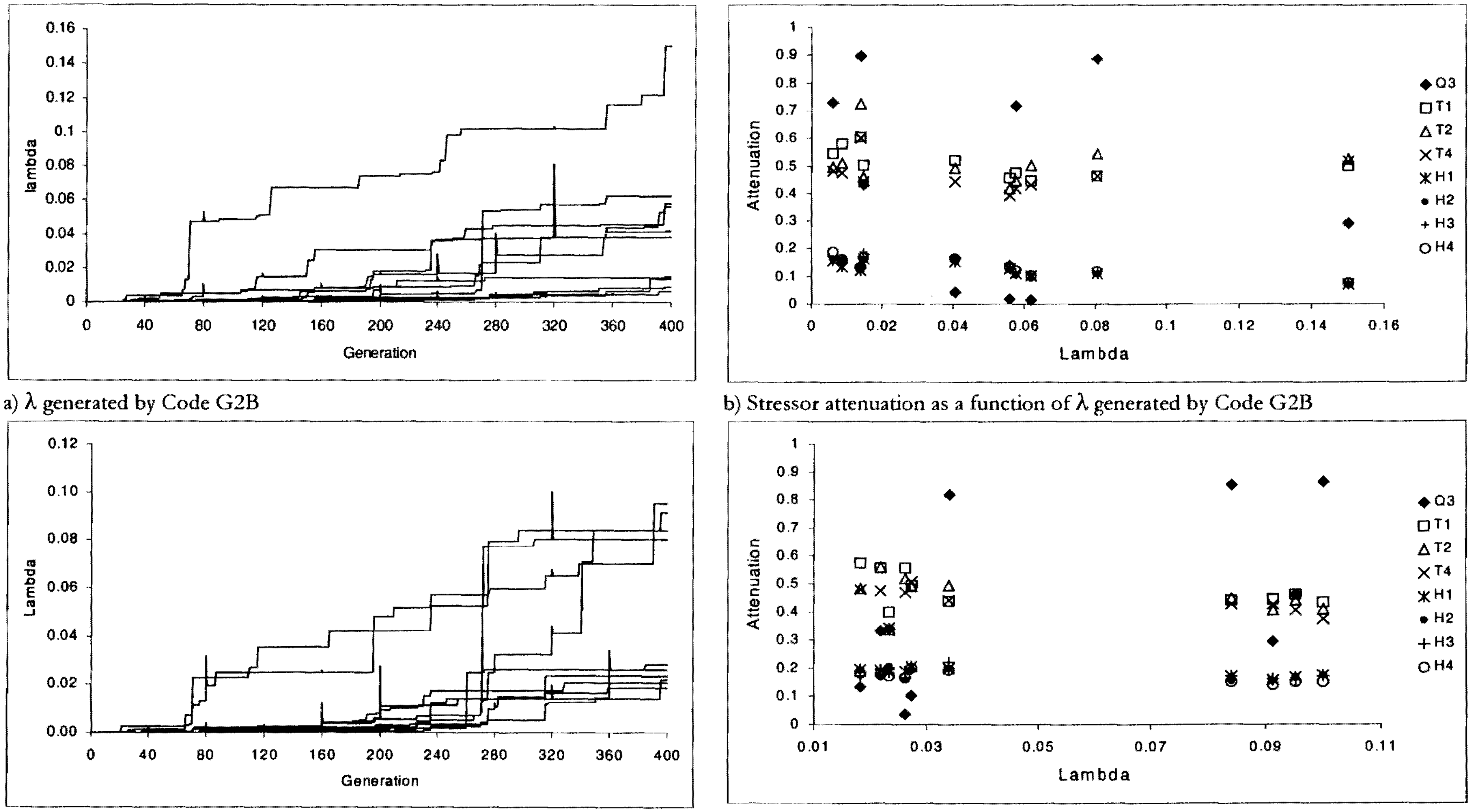


c) λ generated by code G2B



d) Stressor attenuation as a function of λ generated by Code G2B

Figure A4.2 A comparison of the effect of the addition of an equity constraint on the performance of the genetic algorithm.



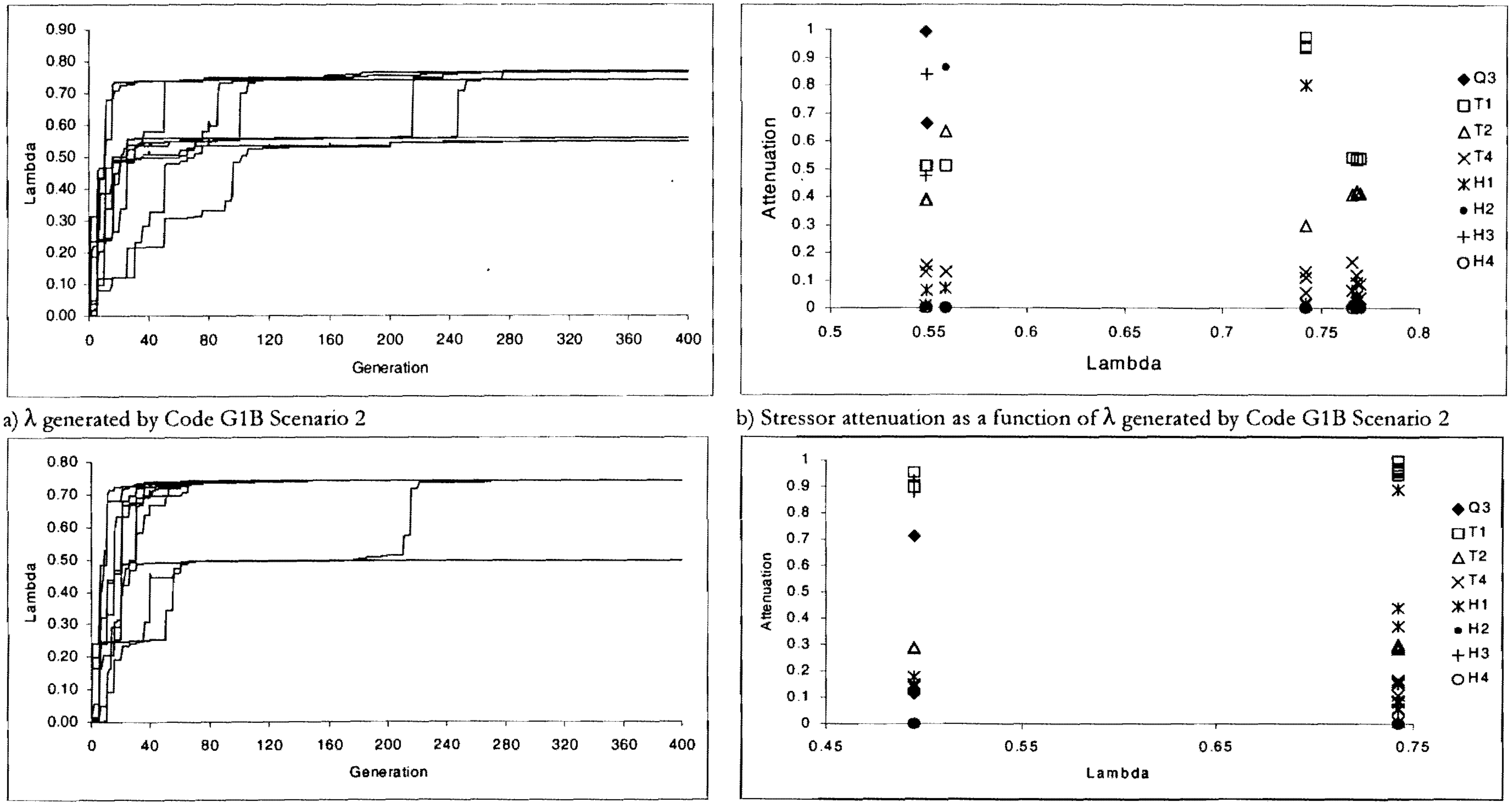
a) λ generated by Code G2B

b) Stressor attenuation as a function of λ generated by Code G2B

c) λ generated by code G3B

d) Stressor attenuation as a function of λ generated by Code G3B

Figure A4.3 A comparison of the effect of the addition of an equity constraint and change to minimum attenuation acceptability on the performance of the genetic algorithm.



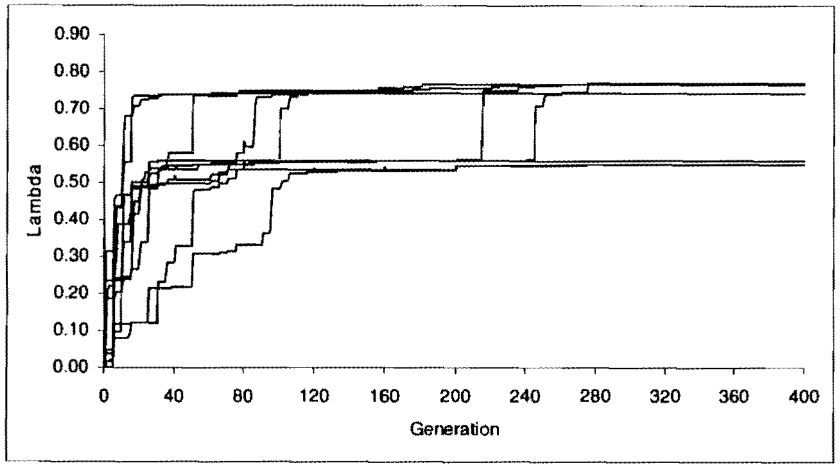
a) λ generated by Code G1B Scenario 2

b) Stressor attenuation as a function of λ generated by Code G1B Scenario 2

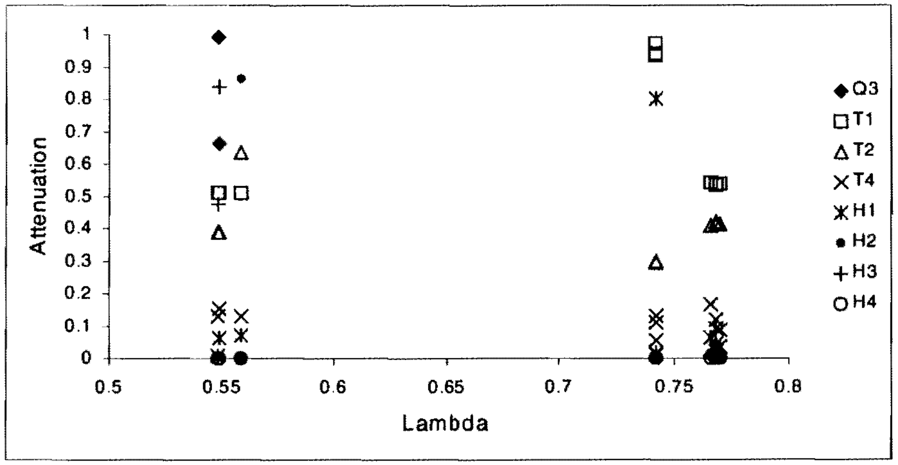
c) λ generated by code G1B Scenario 3

d) Stressor attenuation as a function of λ generated by Code G1B Scenario 3

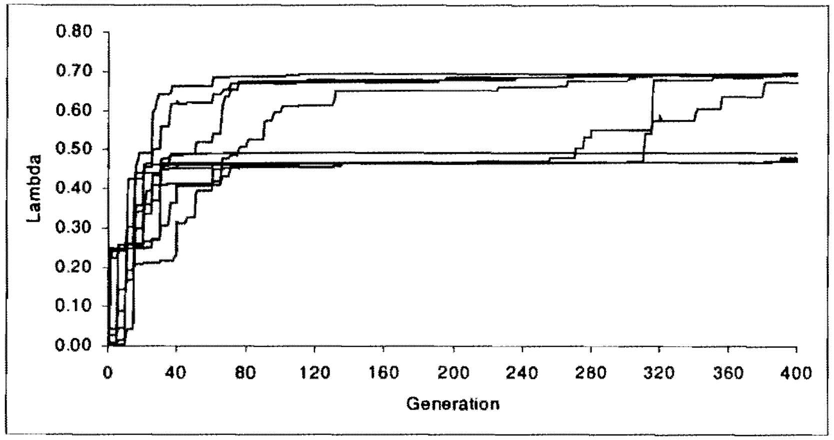
Figure A4.4 A comparison of the effect of a change of attenuation acceptability for toxics at source 1 on the performance of the genetic algorithm.



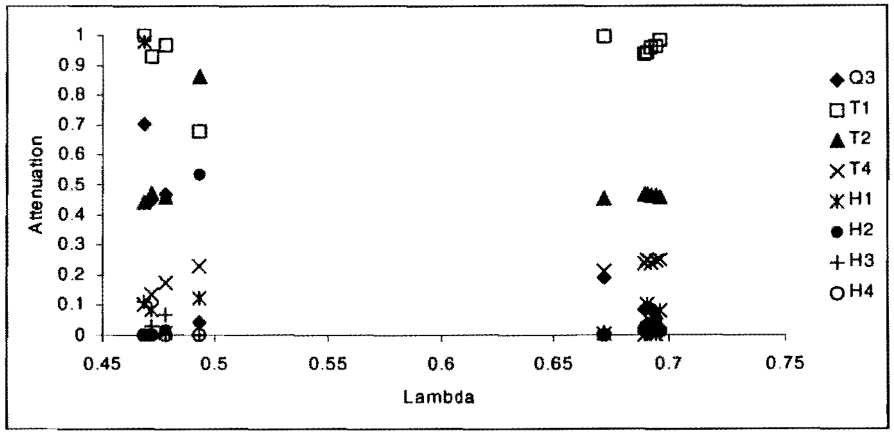
a) λ generated by Code G1B Scenario 2



b) Stressor attenuation as a function of λ generated by Code G1B Scenario 2

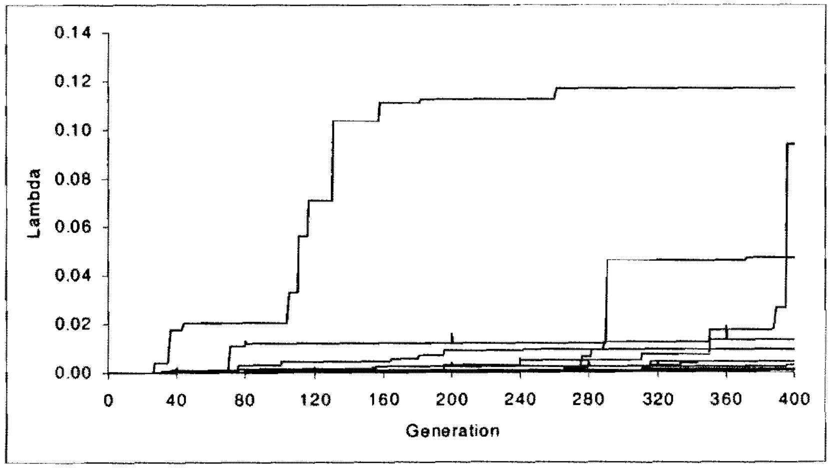


c) λ generated by code G1B Scenario4

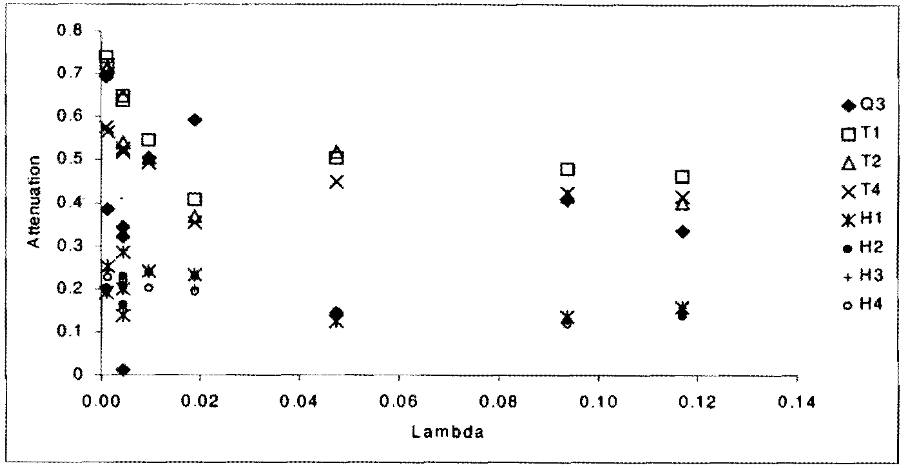


d) Stressor attenuation as a function of λ generated by Code G1B Scenario 4

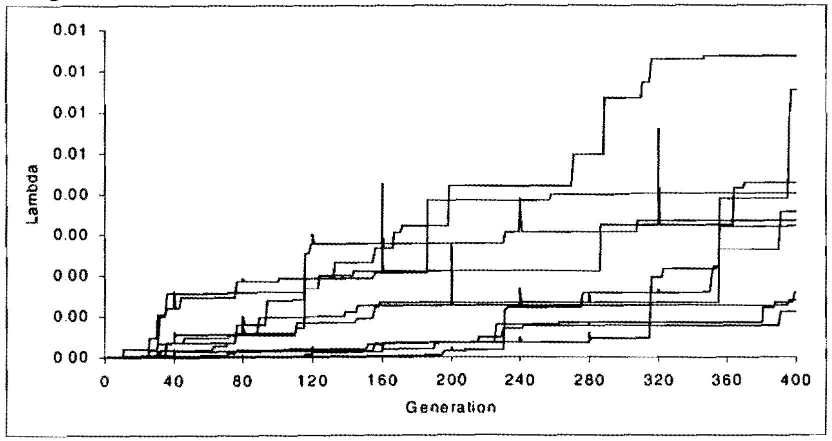
Figure A4.5 A comparison of a change in risk acceptability on the performance of the genetic algorithm.



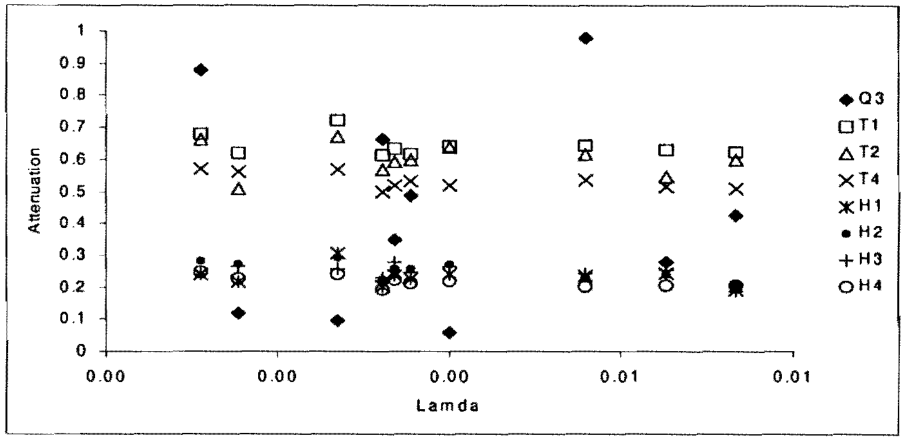
a) λ generated by Code G3B Scenario 3



b) Stressor attenuation as a function of λ generated by Code G3B Scenario 3

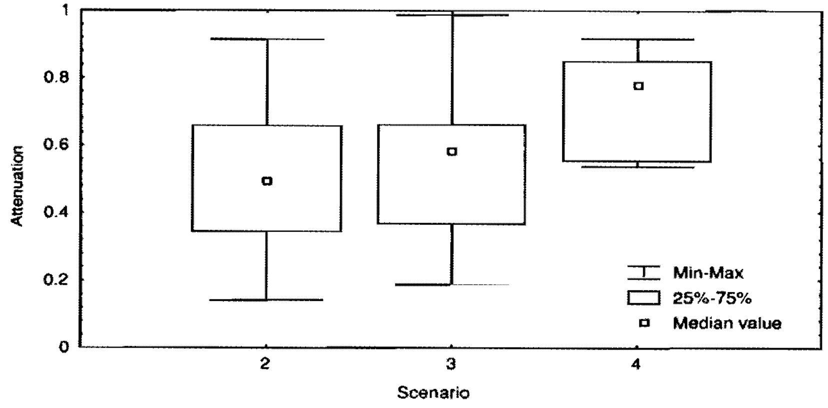


c) λ generated by code G3B Scenario 4

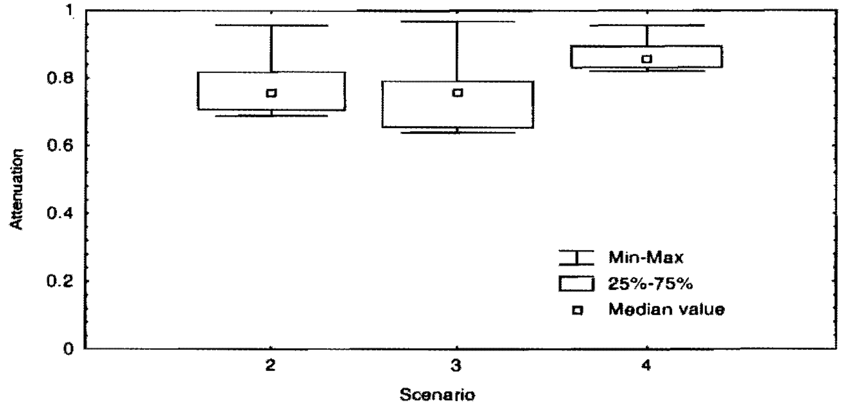


d) Stressor attenuation as a function of λ generated by Code G1B Scenario 4

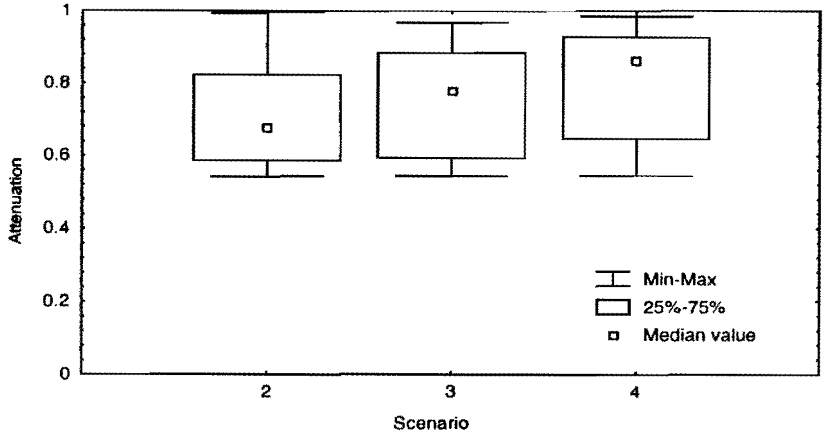
Figure A4.6 A comparison of a change in risk acceptability on the performance of the genetic algorithm with both conjunctive aggregation and equity constraint.



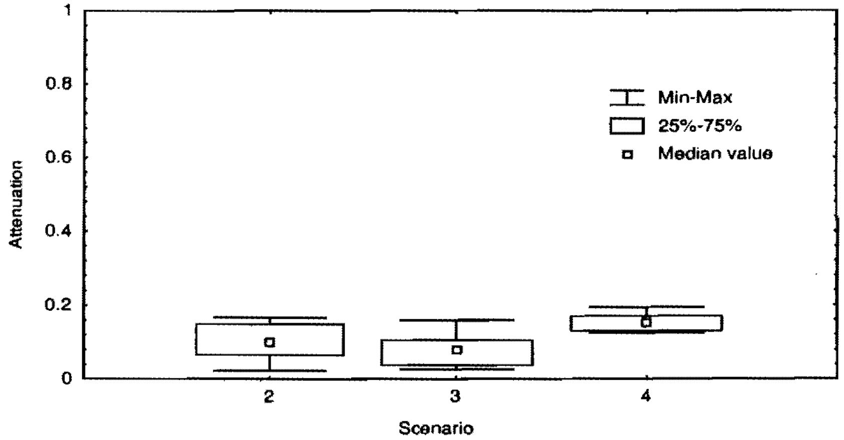
a) xQ3



b) xT1

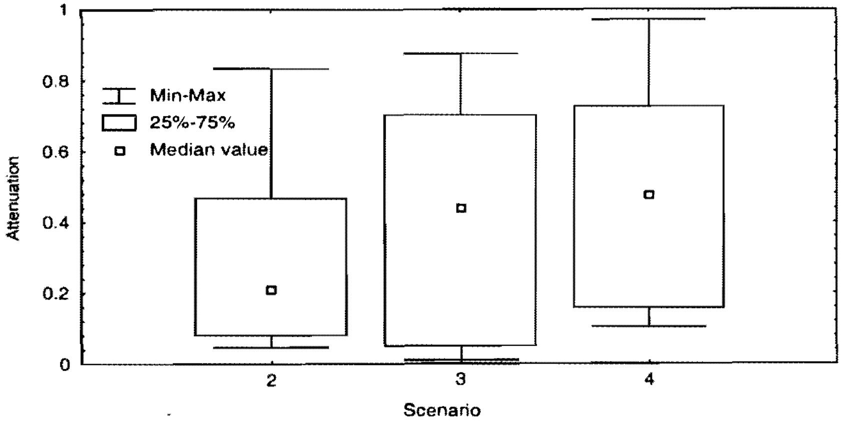


c) xT2

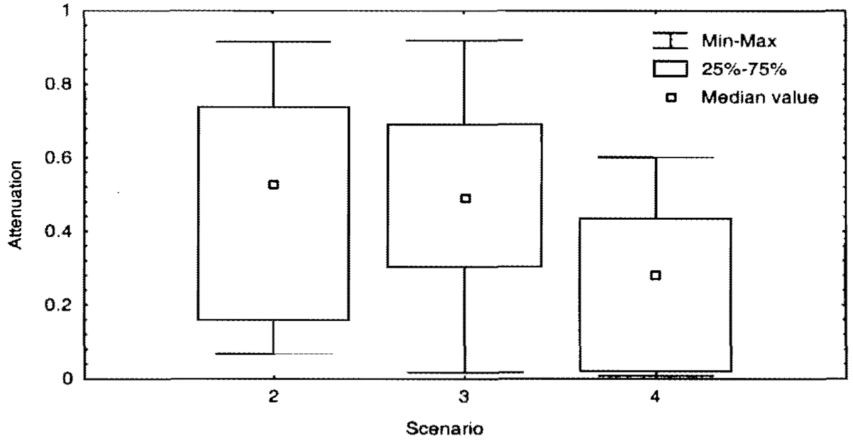


d) xT4

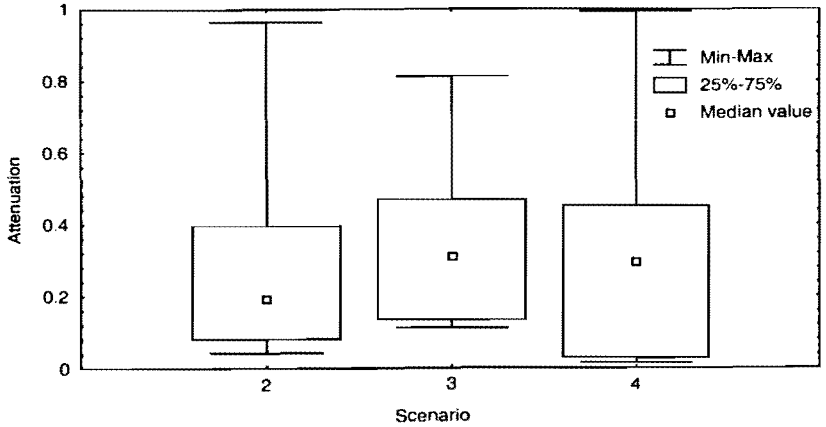
Figure A4.7 The distribution of stressor attenuation values generated by code G4B with degenerate λ = 0.99 values.



e) xH1



f) xH2



g) xH3

Figure A4.6 (continued) The distribution of stressor attenuation values generated by code G4B with degenerate λ = 0.99 values.

A4.3 THE BASIC ALGORITHM CODING G1A IN MS-QBASIC

```
DECLARE SUB climb (t%, m%, x!(), fs!(), pmin!(), pmax!())
DECLARE SUB datainput (infil$, s%, c%, s0!(), f!(), k!(), tau!(), z%(), qm!(), e!(), user!(), regl!())
DECLARE SUB ftox (s%, c%, f!(), x!())
DECLARE SUB findvalue (f!, x!(), t%)
DECLARE SUB initialize (x!(), pmin!(), pmax!(), fs!(), t%, m%)
DECLARE SUB QuickSort (ndim%, SList!(), PList!(), Left%, Right%)
DECLARE SUB encode (x!(), t%, chrom$)
DECLARE SUB offspring (m%, gen%, ch$())
DECLARE SUB decode (chrom$, t%, y!())
DECLARE SUB binadd (x$, y$, z$)
DECLARE SUB binneg (a$, c2$)
DECLARE SUB cvbin (x!, a$)
DECLARE SUB cvdec (y$, y!)
DECLARE SUB Partition (ndim%, SList!(), PList!(), Left%, Right%, part%)
DECLARE SUB calcrisk (i%, j%, muef!(), mus!(), rsk!())
DECLARE SUB value (lamda!, x!())
DECLARE SUB intadd (x!(), y!(), z!())
DECLARE SUB intdiv (x!(), y!(), z!())
DECLARE SUB intinv (x!(), z!())
DECLARE SUB intmult (x!(), y!(), z!())
DECLARE SUB linv (y!, mu!, s!, x!())
DECLARE SUB mueff (i%, j%, e!(), s!(), muef!())
DECLARE SUB mustres (i%, j%, a!, qm!(), st!(), mus!(), poss!)
DECLARE SUB ninv (y!, mu!, s!, x!())
DECLARE SUB satisfy (s%, c%, user!(), regl!(), maxr!, f!(), lamda!)
DECLARE SUB stresdist (a%, s%, c%, p!(), k!(), f!(), tau!(), z%(), s!(), a!)
DECLARE SUB tfnalfa (a!(), alfa!, a1!, a2!)
DECLARE SUB xtof (s%, c%, f!(), x!())
CONST pi = 3.1415926536#
s% = 3  'Number of stressors
c% = 4  'Number of sources
n% = 20 'Number of confidence levels
p% = 10             'epoch number
eps = .0001
CLS
RANDOMIZE TIMER
DIM s0(s%, c% + 1, 3), f(s%, c%), e(s%, 2), k(s%, c%), tau(c% + 1), qm(c%, 2), user(s%, c%, 2),
regl(2), z%(c% + 1), xf(s% * c%)
DEF fnmustepup (min, max, x)
        IF x <= min THEN
                fnmustepup = 0
        ELSEIF x >= max THEN
                fnmustepup = 1
        ELSE
                fnmustepup = (x - min) / (max - min)
        END IF
END DEF
DEF fnmustepdown (min, max, x)
        IF x <= min THEN
                fnmustepdown = 1
        ELSEIF x >= max THEN
                fnmustepdown = 0
        ELSE
                fnmustepdown = (max - x) / (max - min)
        END IF
END DEF
DEF fnsatisfy (x1, x2, x)
```

```
        y1 = .99
        y2 = .01
        k = LOG(y2 * (1 - y1) / ((1 - y2) * y1)) / (x1 - x2)
        ax = EXP(LOG(y1 / (1 - y1)) + k * x1)
                fnsatisfy = ax * EXP(-k * x) / (1 + ax * EXP(-k * x))
END DEF
DEF fntriang (a, b, c, x)
        IF x < a OR x > c THEN
                fntriang = 0
        ELSEIF x <= b THEN
                fntriang = (x - a) / (b - a)
        ELSE
                fntriang = (c - x) / (c - b)
        END IF
END DEF

DEF fnmin (a, b)
        IF a <= b THEN fnmin = a ELSE fnmin = b
END DEF
DEF fnmax (a, b)
        IF a <= b THEN fnmax = b ELSE fnmax = a
END DEF
DEF fnnorm (x, mu, s)
        fnnorm = EXP(-(x - mu) ^ 2 / (2 * s ^ 2)) / (2 * SQR(2 * pi))
END DEF
DEF fnlognorm (x, mu, s)
        fnlognorm = EXP(-(LOG(x) - mu) ^ 2 / (2 * s * SQR(2 * pi))) / (x * s * SQR(2 * pi))
END DEF
'-----------Inputs--------------------------
t% = 8: m% = 2 * t%
DIM x(m%, t%), xi(t%), y(m%, t%), yi(t%), ch$(m%), fs(m%), xb(n%, t%)
DIM oldx(2, t%), lr(m%), lx(m%)
DIM sumxb(t%), xbmax(t%), xbmin(t%), oldxbmax(t%), oldxbmin(t%)
DIM SList(m%), PList(m%, t%)
fil$ = "g1a": f$ = "f.txt": x$ = "x.txt"
idir$ = "c:\data\optin": iex$ = ".dat"
odir$ = "c:\data\"
FOR filecount% = 1 TO 3
        c$ = RIGHT$(STR$(filecount% + 1), 1)
        infil$ = idir$ + c$ + iex$
        outfil1$ = odir$ + fil$ + c$ + f$
        outfil2$ = odir$ + fil$ + c$ + x$
        CALL datainput(infil$, s%, c%, s0(), f!(), k!(), tau!(), z%(), qm(), e!(), user!(), regl())
        CALL ftox(s%, c%, f(), xf())
        m% = 2 * t%
        FOR i% = 1 TO t%
                vbestx(i%) = 0
        NEXT
        REDIM x(m%, t%), xi(t%), y(m%, t%), yi(t%), ch$(m%), fs(m%), xb(n%, t%)
        REDIM oldx(2, t%), lr(m%), lx(m%)
        REDIM sumxb(t%), xbmax(t%), xbmin(t%), oldxbmax(t%), oldxbmin(t%)
        REDIM SList(m%), PList(m%, t%)
        OPEN outfil1$ FOR OUTPUT AS #5
        OPEN outfil2$ FOR APPEND AS #6
'=====================OPTIMIZATION BY GENETIC LGORITHM================
'                 =============MAIN PROGRAMME====================
        try% = 0: scount% = 0
        DO
        try% = try% + 1
        PRINT try%;
```

```
outloop% = 0
vbestf = 1000
FOR j% = 1 TO 2    '--Find 1st two suitable values as parents
        DO
                FOR i% = 1 TO t%
                        xi(i%) = RND
                        x(j%, i%) = xi(i%)
                NEXT
                CALL findvalue(f, xi(), t%)
                fs(j%) = f
        LOOP UNTIL f < 1
NEXT                        '--Arrange 1st 2 values--
IF fs(2) < fs(1) THEN
        SWAP fs(1), fs(2)
        FOR i% = 1 TO t%
                SWAP x(1, i%), x(2, i%)
        NEXT
END IF          '---------------------------
DO
        FOR i% = 1 TO t%
                pmin(i%) = 0
                pmax(i%) = 1
                shift(i%) = 0
                bestx(i%) = 0
        NEXT
        outloop% = outloop% + 1          '--Prepare for epoch---
        count% = 0: gen% = 0: bestf = 100000
        sgen% = 0
        FOR i% = 1 TO t%
                sumxb(i%) = 0
                xbmax(i%) = 0
                xbmin(i%) = 9999
        NEXT
        CALL initialize(x(), pmin(), pmax(), fs(), t%, m%)'----------------------
        CALL QuickSort(t%, fs(), x(), 1, m%)
        FOR i% = 1 TO t%
                x0(i%) = x(1, i%)
        NEXT
        CALL findvalue(f, x0(), t%)
        PRINT #5, scount%;
        FOR i% = 1 TO t%
                PRINT #5, x(1, i%);
        NEXT
        PRINT #5, fs(1); lamdar; lamdax
        DO                          '--Start epoch----------------
                sgen% = sgen% + 1
                count% = count% + 1
                scount% = scount% + 1
                bestf1 = fnmin(bestf, fs(1))
                IF bestf1 < bestf THEN
                        FOR i% = 1 TO t%
                                bestx(i%) = x(1, i%)
                        NEXT
                        bestf = bestf1
                END IF
                FOR i% = 1 TO m%          '---produce chromosomes---
                        FOR j% = 1 TO t%
                                xi(j%) = x(i%, j%)
                        NEXT
                        c$ = ""
```

```
        CALL encode(xi(), t%, c$)
        ch$(i%) = c$
NEXT
CALL offspring(m%, gen%, ch$())  '---do genetic manipulations
FOR i% = 1 TO m%
        c$ = ch$(i%)
        CALL decode(c$, t%, xi())
        FOR j% = 1 TO t%
                x(i%, j%) = xi(j%)
        NEXT
        CALL findvalue(f, xi(), t%)
        fs(i%) = f
NEXT
CALL QuickSort(t%, fs(), x(), 1, m%)
FOR i% = 1 TO t%
        x0(i%) = x(1, i%)
NEXT
CALL findvalue(f, x0(), t%)
PRINT #5, scount%;
FOR i% = 1 TO t%
        PRINT #5, x(1, i%);
NEXT
PRINT #5, fs(1); lamdar; lamdax
IF count% = 1 THEN              '---prepare for next epoch
        FOR i% = 1 TO t%          '--initialise max-min calc params
                oldxbmax(i%) = x(1, i%)
                oldxbmin(i%) = x(1, i%)
        NEXT
ELSE
        FOR i% = 1 TO t%
                oldxbmax(i%) = xbmax(i%)
                oldxbmin(i%) = xbmin(i%)
        NEXT
END IF
IF sgen% < 5 THEN
        FOR i% = 1 TO t%
                sumxb(i%) = sumxb(i%) + x(1, i%)
                newxb = x(1, i%)
                oldxbmax = oldxbmax(i%): oldxbmin = oldxbmin(i%)
                xbmax(i%) = fnmax(oldxbmax, newxb)
                xbmin(i%) = fnmin(oldxbmin, newxb)
        NEXT
ELSE
        FOR i% = 1 TO t%
                sumxb(i%) = sumxb(i%) + x(1, i%)
                rl = 2 * (xbmax(i%) - xbmin(i%))
                IF rl < .4 THEN
                        rl = .4
                ELSEIF rl > .5 THEN
                        rl = .5
                END IF
                pmax(i%) = x(1, i%) + rl * (pmax(i%) - pmin(i%))
                pmin(i%) = x(1, i%) - rl * (pmax(i%) - pmin(i%))
                shift(i%) = ((sumxb(i%) / sgen%) - .5 * (pmax(i%) + pmin(i%))) /
(pmax(i%) - pmin(i%))
                pmax(i%) = pmax(i%) + shift(i%) * (pmax(i%) - pmin(i%))
                IF pmax(i%) >= 1 THEN pmax(i%) = .99999
                pmin(i%) = pmin(i%) + shift(i%) * (pmax(i%) - pmin(i%))
                IF pmin(i%) <= 0 THEN pmin(i%) = .00001
        NEXT
```

```
                        CALL climb(t%, m%, x(), fs(), pmin(), pmax())
                        sgen% = 0
                        FOR i% = 1 TO t%
                              sumxb(i%) = 0
                        NEXT
                  END IF
                  CALL QuickSort(t%, fs(), x(), 1, m%)
            LOOP UNTIL count% = 40
            IF bestf < vbestf THEN
                  FOR i% = 1 TO t%
                        vbestx(i%) = bestx(i%)
                  NEXT
                  vbestf = bestf
                  vbestlr = bestlr: vbestlx = bestlx
            END IF
            FOR i% = 1 TO t%
                  x(1, i%) = bestx(i%)
            NEXT
      LOOP UNTIL outloop% = p%
      PRINT
      FOR i% = 1 TO t%
            PRINT vbestx(i%);
            PRINT #6, vbestx(i%);
      NEXT
      PRINT #6, vbestf
      PRINT vbestf
      LOOP UNTIL try% = 10
      CLOSE #5: CLOSE #6
NEXT 'filecount%
'=======================END OF MAIN PROGRAMME=======================

SUB binadd (x$, y$, z$)
z$ = "": co = 0
FOR i% = 16 TO 1 STEP -1
      a = VAL(MID$(x$, i%, 1)): b = VAL(MID$(y$, i%, 1))
      c = a + b + co
      IF c >= 2 THEN
            d = 2 - c
            co = 1
      ELSE
            d = c
            co = 0
      END IF
      z$ = RIGHT$(STR$(d), 1) + z$
NEXT
END SUB

SUB binneg (a$, c2$)
CALL cvbin(1, one$)
FOR i% = 1 TO 15
      one$ = "0" + one$
NEXT
c$ = ""
FOR i% = 1 TO 16
      IF MID$(a$, i%, 1) = "1" THEN
            c$ = c$ + "0"
      ELSEIF MID$(a$, i%, 1) = "0" THEN
            c$ = c$ + "1"
      END IF
NEXT
```

```
CALL binadd(c$, one$, c2$)
END SUB

SUB calcrisk (i%, j%, muef(), mus(), rsk())
        maxr = 0: mx = 0
        muef1 = muef(i%, j%, 1): muef2 = muef(i%, j%, 2)
        mus1 = mus(i%, j%, 1): mus2 = mus(i%, j%, 2)
        rsk(i%, j%, 1) = fnmin(muef1, mus1)
        rsk(i%, j%, 2) = fnmin(muef2, mus2)
END SUB

SUB climb (t%, m%, x(), fs(), pmin(), pmax())
DIM range(t%), xi(t%)
        FOR i% = 1 TO t%
                range(i%) = pmax(i%) - pmin(i%)
        NEXT
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        x(i%, j%) = RND * range(j%) + pmin(j%)
                        xi(j%) = x(i%, j%)
                NEXT
                CALL findvalue(f, xi(), t%)
                fs(i%) = f
        NEXT
END SUB

SUB cvbin (x, a$)
a$ = ""
IF x >= 0 THEN
        xa = x
        FOR i% = 16 TO 1 STEP -1
                a = 2 ^ (i% - 1)
                IF a > xa THEN
                        p$ = "0"
                ELSE
                        p$ = "1"
                        xa = xa - a
                END IF
                a$ = a$ + p$
        NEXT
ELSE
        y = -x
        ya = y
        FOR i% = 16 TO 1 STEP -1
                a = 2 ^ (i% - 1)
                IF a > ya THEN
                        p$ = "0"
                ELSE
                        p$ = "1"
                        ya = ya - a
                END IF
                a$ = a$ + p$
        NEXT
        CALL binneg(a$, c$)
        a$ = c$
END IF
END SUB

SUB cvdec (y$, y)
        y = 0
```

```
        FOR i% = 1 TO 16
                y = y + VAL(MID$(y$, i%, 1)) * 2 ^ (16 - i%)
        NEXT
END SUB

SUB datainput (infil$, s%, c%, s0!(), f!(), k!(), tau!(), z%(), qm(), e!(), user!(), regl())
OPEN infil$ FOR INPUT AS #1
ct% = 0
FOR i% = 1 TO s%
        FOR k% = 1 TO 3
                FOR j% = 0 TO c%
                        INPUT #1, s0(i%, j%, k%): ct% = ct% + 1
                NEXT
        NEXT
NEXT
FOR j% = 1 TO 2
        FOR i% = 1 TO c%
                INPUT #1, qm(i%, j%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO s%
        FOR k% = 1 TO 2
                FOR j% = 1 TO c%
                        INPUT #1, e(i%, k%): ct% = ct% + 1
                NEXT
        NEXT
NEXT
FOR j% = 1 TO s%
        FOR i% = 1 TO c%
                INPUT #1, f(j%, i%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO c%
        INPUT #1, z%(i%): ct% = ct% + 1
NEXT
FOR j% = 1 TO s%
        FOR i% = 1 TO c%
                INPUT #1, k(j%, i%): ct% = ct% + 1
        NEXT
NEXT
FOR i% = 1 TO c%
        INPUT #1, tau(i%): ct% = ct% + 1
NEXT
FOR k% = 1 TO s%
FOR j% = 1 TO 2
        FOR i% = 1 TO c%
                INPUT #1, user(k%, i%, j%): ct% = ct% + 1
        NEXT
NEXT
NEXT
INPUT #1, regl(1)
INPUT #1, regl(2)
CLOSE #1
END SUB

SUB decode (chrom$, t%, y())
'-----------------------------Decode chromosome----------------
DIM y$(t%)
FOR i% = 1 TO t%
```

```
        p% = 1 + 16 * (i% - 1)
        y$(i%) = MID$(chrom$, p%, 16)
        y$ = y$(i%)
        IF VAL(LEFT$(y$, 1)) = 1 THEN
                CALL binneg(y$, y1$)
                CALL cvdec(y1$, y)
                y = -y
        ELSE
                CALL cvdec(y$, y)
        END IF
        y(i%) = y / 1000
NEXT
END SUB

SUB discrete (snum%, num%, xlow, xup, x())
FOR i% = 1 TO num%
        x(snum%, i%) = xlow + (i% - 1) * (xup - xlow) / (num% - 1)
NEXT
END SUB

SUB encode (x(), t%, chrom$)
'----------------------------Encode chromosome; 3 decimal accuracy---
chrom$ = ""
FOR i% = 1 TO t%
        x = x(i%) * 1000
        CALL cvbin(x, a$)
        chrom$ = chrom$ + a$
NEXT
END SUB

SUB findvalue (f, x(), t%)
        SHARED s%, c%
        er% = 0
        FOR i% = 1 TO t%
                IF x(i%) < 0 OR x(i%) > 1 THEN er% = 1
        NEXT
        IF er% = 0 THEN
                CALL value(lamda, x())
                f = 1 - lamda
        ELSE
                f = 101010
        END IF
END SUB

SUB ftox (s%, c%, f(), x())
SHARED z%(), t%
k% = 0
FOR i% = 1 TO s%
        FOR j% = 1 TO c%
                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) = 0) OR i% = s%
THEN
                k% = k% + 1
                x(k%) = f(i%, j%)
                END IF
        NEXT
NEXT
IF t% <> k% THEN t% = k%
END SUB

SUB initialize (x(), pmin(), pmax(), fs(), t%, m%)
```

```
'----------------------------------------Initialize variables-----
        DIM xi(t%)
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        pwr% = INT(RND * 2) + 1
                        x(i%, j%) = x(1, j%) + (-1) ^ pwr% * RND * .5 * (pmax(j%) - pmin(j%))
                        xi(j%) = x(i%, j%)
                NEXT
                        CALL findvalue(f, xi(), t%)
                        fs(i%) = f
        NEXT
END SUB

SUB intadd (x(), y(), z())
z(1) = x(1) + y(1)
z(2) = x(2) + y(2)
END SUB

SUB intdiv (x(), y(), z())
DIM a(2)
CALL intinv(y(), a())
temp = a(1)
a(1) = a(2)
a(2) = temp
CALL intmult(x(), a(), z())
END SUB

SUB intinv (x(), z())
z(1) = 1 / x(2): z(2) = 1 / x(1)
END SUB

SUB intmult (x(), y(), z())
'a = x(1) * y(1): b = x(1) * y(2): c = x(2) * y(1): d = x(2) * y(2)
z(1) = x(1) * y(1)'fnmin(d, fnmin(c, fnmin(a, b)))
z(2) = x(2) * y(2)'fnmax(d, fnmax(c, fnmax(a, b)))
END SUB

SUB intsub (x(), y(), z())
z(1) = x(1) - y(2)
z(2) = x(2) - y(1)
END SUB

SUB linv (y, mu, s, x())
        y1 = y * .99999 / SQR(2 * pi)
        zpos = SQR(-2 * LOG(ABS(SQR(2 * pi) * y1)))
        zneg = -SQR(-2 * LOG(ABS(SQR(2 * pi) * y1)))
        x(1) = EXP(mu1 + s * zneg): x(2) = EXP(mu1 + s * zpos)
END SUB

SUB mueff (i%, j%, e(), s(), muef())
        IF i% = 1 THEN
                a = .2: b = .8
                q01 = s(1, 0, 1): IF q01 = 0 THEN q01 = .0001
                q02 = s(1, 0, 2): IF q02 = 0 THEN q02 = .0001
                x1 = (q01 - s(1, j%, 1)) / q01: x2 = (q02 - s(1, j%, 2)) / q02
        ELSE
                a = e(i%, 1): b = e(i%, 2)
                x1 = s(i%, j%, 1): x2 = s(i%, j%, 2)
        END IF
        e1 = fnmustepup(a, b, x1)
```

```
        e2 = fnmustepup(a, b, x2)
        muef(i%, j%, 1) = e1
        muef(i%, j%, 2) = e2
END SUB

SUB mustres (i%, j%, a, qm(), st(), mus(), poss)
                mus(i%, j%, 1) = a: mus(i%, j%, 2) = a: poss = a
END SUB

SUB ninv (y, mu, s, x())
        y1 = y / SQR(2 * pi)
        zpos = SQR(-2 * s ^ 2 * LOG(y))
        zneg = -SQR(-2 * s ^ 2 * LOG(y))
        x(1) = mu + s * zneg: x(2) = mu + s * zpos
END SUB

SUB offspring (m%, gen%, ch$())
'----------------Produce offspring-----
SHARED t%
DIM f2$(m%)
lamda = 1
'---select parents
FOR i% = 5 TO m%
DO
        pno1% = INT(-LOG(1 - RND) / lamda + 1)
        pno2% = INT(-LOG(1 - RND) / lamda + 1)
LOOP UNTIL pno1% <> pno2% AND pno1% < m% AND pno2% < m%
        f2$(i%) = ""
        dch$ = ""
        FOR j% = 1 TO t%
                gen% = gen% + 1
                byte% = 16 * (j% - 1) + 1
                slct% = INT(RND * 2)    'randomly select parent 1 or 2
                                IF slct% = 1 THEN
                        a$ = MID$(ch$(pno1%), byte%, 16)
                ELSE
                        a$ = MID$(ch$(pno2%), byte%, 16)
                END IF
                IF gen% = 10 THEN
                        mubit1% = INT(RND * 16) + 1: mubit2% = INT(RND * 16) + 1
                        dummy$ = ""
                        FOR k% = 1 TO 16
                                IF k% <> mubit1% OR k% <> mubit2% THEN
                                        dummy$ = dummy$ + MID$(a$, k%, 1)
                                ELSE
                                IF MID$(a$, k%, 1) = "1" THEN
                                        dummy$ = dummy$ + "0"
                                ELSE
                                        dummy$ = dummy$ + "1"
                                END IF
                                END IF
                        NEXT
                        gen% = 0
                        a$ = dummy$
                END IF
                dch$ = dch$ + a$
        NEXT j%
        f2$(i%) = dch$
NEXT i%
FOR i% = 5 TO m%
```

```
        ch$(i%) = f2$(i%)
NEXT
END SUB

SUB Partition (ndim%, SList(), PList(), Left%, Right%, part%)
        DIM temp(ndim%)
        v = SList(Right%)
        indx% = Left% - 1
        Jndx% = Right%
         DO
                DO
                 indx% = indx% + 1
                LOOP UNTIL SList(indx%) >= v
                DO
                 Jndx% = Jndx% - 1
                LOOP UNTIL SList(Jndx%) <= v
                temp = SList(indx%)
                SList(indx%) = SList(Jndx%)
                SList(Jndx%) = temp
                FOR i% = 1 TO ndim%
                temp(i%) = PList(indx%, i%)
                PList(indx%, i%) = PList(Jndx%, i%)
                PList(Jndx%, i%) = temp(i%)
                NEXT
         LOOP UNTIL Jndx% <= indx%
        SList(Jndx%) = SList(indx%)
        SList(indx%) = SList(Right%)
        SList(Right%) = temp
         FOR i% = 1 TO ndim%
                PList(Jndx%, i%) = PList(indx%, i%)
                PList(indx%, i%) = PList(Right%, i%)
                PList(Right%, i%) = temp(i%)
         NEXT
         part% = indx%
END SUB

SUB QuickSort (ndim%, SList(), PList(), Left%, Right%)
        IF Left% <= Right% THEN
                CALL Partition(ndim%, SList(), PList(), Left%, Right%, indx%)
                CALL QuickSort(ndim%, SList(), PList(), Left%, indx% - 1)
                CALL QuickSort(ndim%, SList(), PList(), indx% + 1, Right%)
        END IF
END SUB

SUB satisfy (s%, c%, user(), regl(), maxr, f(), lamda)
SHARED t%, z%(), lamdar, lamdax
min = regl(1): max = regl(2)
lamdar = fnsatisfy(min, max, maxr)
'---calculate user satisfaction---
lmdx = 0
FOR i% = 1 TO c%
        lamdai = 1
        FOR j% = 1 TO s%
                IF (j% = 1 AND z%(i%) = 1) OR (j% > 1 AND j% < s% AND z%(i%) = 0) OR j% = s%
                THEN
                        min = user(j%, i%, 1): max = user(j%, i%, 2)
                        v = f(j%, i%)
                        lx = fnsatisfy(min, max, v)
                        lamdai = fnmin(lx, lamdai)
                END IF
```

```
        NEXT
        lmdx = lmdx + lamdai
NEXT
lamdax = lmdx / c%
lamda = fnmin(lamdar, lamdax)
'PRINT lamdar; lamdax,
END SUB

SUB stresdist (a%, s%, c%, p(), k(), f(), tau(), z%(), s(), a)
        SHARED n%
        DIM s0(s%, c%, 2), s1(s%, c%, 2), sv(2), qu(2), qi(2), z(2)
        DIM su(2), si(2), lu(2), li(2), qt(2), lt(2), tri(3)
        tau(0) = 0
        tau = 0
        IF a = 0 THEN a = .01
        FOR j% = 1 TO s% - 1
                mu = p(j%, 0, 1): s = p(j%, 0, 2)
                IF j% = 1 THEN
                        mu = LOG(mu)
                        CALL linv(a, mu, s, sv())
                ELSE
                        CALL ninv(a, mu, s, sv())
                END IF
                s(j%, 0, 1) = sv(1): s(j%, 0, 2) = sv(2)
                s0(j%, 0, 1) = sv(1): s0(j%, 0, 2) = sv(2)
                s1(j%, 0, 1) = sv(1): s1(j%, 0, 2) = sv(2)
        NEXT
        FOR src% = 1 TO c%
                mu = p(1, src%, 1): s = p(1, src%, 2)
                CALL linv(a, mu, s, sv())
                s0(1, src%, 1) = sv(1)
                s0(1, src%, 2) = sv(2)
                tau = tau(src%) + tau
                f = (1 - f(1, src%))
                s1(1, src%, 1) = s0(1, src%, 1) * (-f) ^ z%(src%)
                s1(1, src%, 2) = s0(1, src%, 2) * (-f) ^ z%(src%)
                qu(1) = s(1, src% - 1, 1)
                qu(2) = s(1, src% - 1, 2)
                qi(1) = s1(1, src%, 1)
                qi(2) = s1(1, src%, 2)
                FOR stres% = 2 TO s% - 1
                        degfactor = EXP(-k(stres%, src%) * tau)
                        mu = p(stres%, src%, 1): s = p(stres%, src%, 2)
                        CALL ninv(a, mu, s, sv())
                        s0(stres%, src%, 1) = sv(1)
                        s0(stres%, src%, 2) = sv(2)
                        f = (1 - f(stres%, src%))
                        s1(stres%, src%, 1) = f * (1 - z%(src%)) * s0(stres%, src%, 1) + z%(src%) *
        degfactor * s(stres%, src% - 1, 1)
                        s1(stres%, src%, 2) = f * (1 - z%(src%)) * s0(stres%, src%, 2) + z%(src%) *
        degfactor * s(stres%, src% - 1, 2)
                        su(1) = s(stres%, src% - 1, 1) * degfactor
                        su(2) = s(stres%, src% - 1, 2) * degfactor
                        si(1) = s1(stres%, src%, 1)
                        si(2) = s1(stres%; src%, 2)
                        CALL intmult(su(), qu(), lu())
                        CALL intmult(si(), qi(), li())
                        CALL intadd(lu(), li(), lt())
                        CALL intadd(qu(), qi(), qt())
                        CALL intdiv(lt(), qt(), z())
```

```
                IF z%(src%) = 0 THEN
                        s(stres%, src%, 1) = z(1): s(stres%, src%, 2) = z(2)
                ELSE
                        s(stres%, src%, 1) = su(1): s(stres%, src%, 2) = su(2)
                END IF
                s(1, src%, 1) = qt(1): s(1, src%, 2) = qt(2)
        NEXT 'stressor
    NEXT 'source
    FOR src% = 1 TO c%
        f = (1 - f(3, src%))
        tri(1) = p(3, src%, 1) * f
        tri(2) = p(3, src%, 2) * f
        tri(3) = p(3, src%, 3) * f
        CALL tfnalfa(tri(), a, a1, a2)
        s(s%, src%, 1) = a1
        s(s%, src%, 2) = a2
    NEXT
END SUB

SUB tfnalfa (a(), alfa, a1, a2)
    a1 = a(1) + alfa * (a(2) - a(1))
    a2 = a(3) - alfa * (a(3) - a(2))
END SUB

SUB trinv (alpha, a, b, c, x())
    x(1) = alpha * (b - a) - a
    x(2) = c - alpha * (c - b)
END SUB

SUB value (lamda, x())
    SHARED s%, c%, n%, a%, s0!(), f(), k!(), tau!(), z%(), qm(), e!(), user!(), regl()
    DIM min(s%), max(s%), st(s%, c% + 1, 3), mus(s%, c%, 2), muef(s%, c%, 2)
    DIM r(s%, c%, 2)
    CALL xtof(s%, c%, f(), x())
    FOR a% = 0 TO n%
        a = a% / n%
        'PRINT #2, a; : PRINT #3, a; : PRINT #4, a;
        CALL stresdist(a%, s%, c%, s0(), k(), f(), tau(), z%(), st(), a)
        maxr = 0: minr = 0
        FOR j% = 1 TO c%
            mxr = 0: mnr = 0
            FOR i% = 1 TO s%
                CALL mustres(i%, j%, a, qm(), st(), mus(), poss)
                CALL mueff(i%, j%, e(), st(), muef())
                CALL calcrisk(i%, j%, muef(), mus(), r())
                hrsk = r(i%, j%, 2)
                lrsk = r(i%, j%, 1)
                mxr = fnmax(mxr, hrsk): mnr = fnmax(mnr, lrsk)
            NEXT
            maxr = fnmax(maxr, fnmax(mxr, mnr))
        NEXT
        'PRINT #2, " ": PRINT #3, " ": PRINT #4, " "
    NEXT
    CALL satisfy(s%, c%, user(), regl(), maxr, f(), lamda)
    CALL ftox(s%, c%, f(), x())
END SUB

SUB xtof (s%, c%, f(), x())
SHARED z%()
k% = 0
```

```
FOR i% = 1 TO s%
        FOR j% = 1 TO c%
                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) = 0) OR i% = s%
THEN
                        k% = k% + 1
                        f(i%, j%) = x(k%)
                ELSE
                        f(i%, j%) = 0
                END IF
        NEXT
NEXT
END SUB
```

A4.3.1 INITIALISATION FROM AN EXPONENTIAL DISTRIBUTION: REPLACEMENT FOR SUB INITIALISE

```
SUB initialize (x(), pmin(), pmax(), fs(), t%, m%)
'-------------Initialize variables (EXP distr)-----
        DIM xi(t%)
        FOR i% = 3 TO m%
                FOR j% = 1 TO t%
                        a = pmin(j%): b = pmax(j%)
                        mu = .5 * (b - a)
                        l = .69314718# / mu
                        x(i%, j%) = -LOG(1 - RND * (b - a)) / l
                        xi(j%) = x(i%, j%)
                NEXT
                        CALL findvalue(f, xi(), t%)
                        fs(i%) = f
        NEXT
END SUB
```

A4.3.2 ADDING AN EQUITY CONSTRAINT: REPLACEMENT FOR SUB FINDVALUE

```
SUB findvalue (f, x(), t%)
        SHARED s%, c%, z%(), leqmin
        er% = 0
        FOR i% = 1 TO t%
                IF x(i%) < 0 OR x(i%) > 1 THEN er% = 1
        NEXT
        IF er% = 0 THEN
                CALL value(lamda, x())
                k% = 0
                leqmin = 10
                FOR i% = 1 TO s%
                        min = 10: max = 0
                        FOR j% = 1 TO c%
                                IF (i% = 1 AND z%(j%) = 1) OR (i% > 1 AND i% < s% AND z%(j%) =
0) OR i% = s% THEN
                                        k% = k% + 1
                                        x1 = x(k%)
                                        min = fnmin(x1, min): max = fnmax(x1, max)
                                END IF
                        NEXT
                        IF min + max > 0 AND min < 1 AND max < 1 THEN
                                dx = ABS(min - max) * 2 / (min + max)
                                leq = fnsatisfy(.01, .2, dx)
```

```
                ELSE
                    leq = 0
                END IF
                leqmin = fnmin(leqmin, leq)
            NEXT
            lamda = fnmin(lamda, leqmin)
            f = 1 - lamda
        ELSE
            f = 101010
        END IF
END SUB
```

4.3.3 CHANGING TO THE CONJUNCTION OPERATOR FOR λ_X : REPLACEMENT FOR SUB SATISFY

```
SUB satisfy (s%, c%, user(), regl(), maxr, f(), lamda)
SHARED t%, z%(), lamdax, lamdar
min = regl(1): max = regl(2)
lamdar = fnsatisfy(min, max, maxr)
'---calculate user satisfaction---
lmdx = 0
FOR i% = 1 TO c%
    lamdai = 100: lamdax = 100
    FOR j% = 1 TO s%
        IF (j% = 1 AND z%(i%) = 1) OR (j% > 1 AND j% < s% AND z%(i%) = 0) OR j%
= s% THEN
            min = user(j%, i%, 1): max = user(j%, i%, 2)
            v = f(j%, i%)
            lx = fnsatisfy(min, max, v)
            lamdai = fnmin(lx, lamdai)
        END IF
    NEXT
    lamdax = fnmin(lamdax, lamdai)
NEXT
lamda = fnmin(lamdar, lamdax)
END SUB
```

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Structure

The document is presented in three Parts:

Part 1: Presents the background and an overview of the work done as well as the main conclusions.

Part 2: Presents the more detailed technical aspects of the work, such as the background to the papers and supplementary information pertaining to the methodology and results reported in the papers.

Part 3: (This Part) Presents some of the papers that have been published in peer reviewed literature and that are included for quick reference.

Part 3:

Technical Papers

Rationale for an ecological risk approach for South African water resource management

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Abstract

The principle of ecosystem protection in the South African Water Act requires that water resource management tools for a multiple stressor environment be tailored to the characteristics of the aquatic ecosystem. The requirements of the Act, the characteristics of aquatic ecosystems as well as co-occurrence of diverse stressors are considered. Although single substance criteria have a useful role, they are not sufficient for resource management within the context of the ecological reserve. It is proposed that an effect-likelihood approach has the potential to address the variability and uncertainty in management of a surface water body subject to multiple stressors. An in-stream receiving water risk objective approach might be considered.

Glossary

ERA	Ecological risk assessment
Hazardous	Having the potential to cause an (undesired) effect.
IFR	In-stream flow requirement
SAWQG	South African Water Quality Guidelines
Stressor	An anthropogenic substance, form of energy or circumstance that may cause a loss of sustainable ecosystem function.

Introduction

The South African national water policy considers the aquatic ecosystem to be an integral part of the resource base from which water is derived for human and environmental use, but "only that water required to meet basic human needs and maintain environmental sustainability will be guaranteed as a right. This will be known as the Reserve" (DWAF, 1997). This concept was also embodied in the National Water Act (NWA, 1998). The environmental or ecological aspect of the reserve has been identified in such a way that it must ensure water quantity and water quality which are appropriate to meet these needs. The term resource quality "is used to include the health of all parts of the water resource, which together make up an 'ecosystem', including plant and animal communities and their habitats" (DWAF, 1997).

This paper presents a rationale for the use of ecological risk in water resource management in South Africa within the context of the NWA.

Background

Two distinct philosophical approaches that can be applied to water resource quality management are summarised in Table 1.

While the approaches in Table 1 are presented as extremes in philosophy, there is a growing appreciation for the need for, and a movement toward, a holistic, integrative approach in environmental management generally and water resource management in particular (e.g. Foran and Fink, 1993; EEC, 1994; Schneiders, et al., 1996; USEPA, 1997). Such a holistic approach to water resource management strongly features sustainability linked to some ecological entity (or objective) (e.g. CUWVO, 1988; Wils et al., 1994; Schneiders et al., 1995; USEPA, 1997). The ecological objectives then become either directly or indirectly the basis of, for example, water quality criteria. Ecological risk methodology can be applied to both extremes and an integrated approach and does not stand in contrast to any of these approaches.

A proposal for the application of ecological risk to the ecological reserve is shown in Fig. 1. The rationale of using ecological risk concepts in water resource management is based on three observations:

- the implications of aspects of the NWA as indicated above,
- the "diverse stressor problem" and
- the inherent characteristics of aquatic ecosystems.

Implications of the NWA

It is implicitly recognised that use of the resource is not only allowed, but is also necessary for the well-being of the country and that this use needs to be managed in a way that will ensure sustainability. In this context it is noted that:

- The terms "use" refers not only to consumption and recreational use, but also to discharge of anything that may affect, *inter alia*, the sustainability of use.
- The NWA makes provision for protective measures for the water resource which includes classification of the resource and setting resource quality objectives that will give effect to the reserve set for that class.
- The ecological component of the reserve refers to a quantity and quality of water that will ensure ecologically sustainable development of the resource.
- Resource quality includes the quantity, pattern, timing, water level and assurance of in-stream flow, the physical, chemical and biological characteristics of the water, the character and condition of the in-stream and riparian habitat as well as the characteristics, condition and distribution of the aquatic biota.

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TABLE 1
A comparison of a technology-based and an ecological effect-based approach to resource management

Aspect	Technology-based approach	Ecological effect-based approach
Point of departure	Technology determines the best attainable stressor levels.	Ecological effect determines the most suitable stressor levels
Characteristic expressions	Best available technology (BAT); Best available technology not entailing excessive cost (BATNEEC); Best management practice (BMP); Best practical technology (BPT), etc.	"Fishable and swimable rivers"; "protecting most species most of the time", "maintaining sustainable ecological function", etc.
Main advantage	Proven technological feasibility.	Directly related to environmental goals
Main disadvantage	Environmental impact largely retrospective.	Required stressor levels not necessarily feasible or viable.

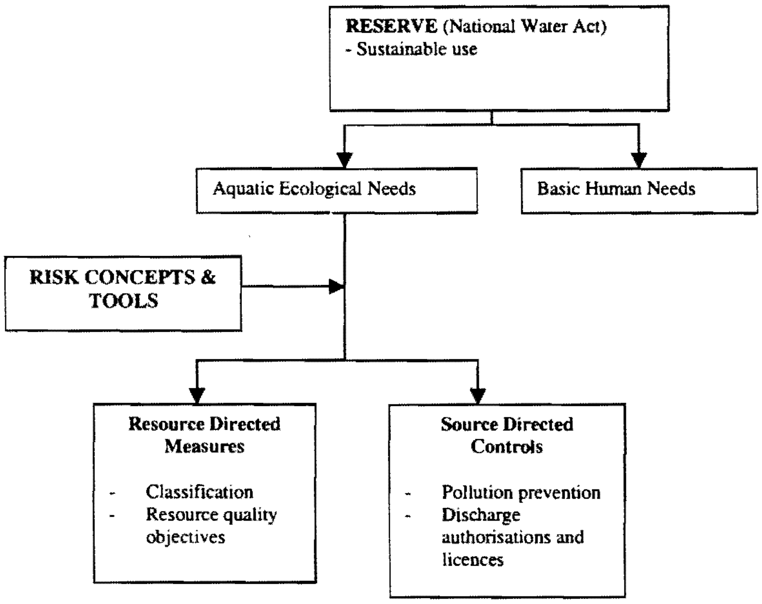

Figure 1
The potential inputs of ecological risk methodology to aspects of water resource quality management.

It is recognised that some activities that may cause stress to the aquatic ecosystem will have to be allowed, but that these have to be controlled in a manner that allows ecological sustainability.

Furthermore, the NWA differentiates between classes of resources, which correspond to a differentiation in some aspect of sustainability. Risk to the resource base was proposed as the basis of differentiation (DWAF, 1997). Here, irreversible damage to the resource base approximates a loss of sustainability.

Consequently, although the term "risk" does not appear explicitly in the NWA as the basis for classification, implicitly it is recognised that different classes of a resource will be subject to different degrees of risk of unsustainability and, by implication, different activities will result in different levels of risk.

The diverse stressor problem

Water use may entail a change in resource characteristics such as chemical composition, physical characteristics, flow and water depth (in the case of rivers), habitat for aquatic organisms, etc. The variables by which these characteristics are measured could conceivably reach a point where it has the potential to cause harm to the aquatic ecosystem.

Definition of a stressor

A stressor could be any substance or circumstance related to the aquatic environment, which could cause the aquatic ecosystem to lose sustainable ecological function. A pollutant would, by definition, be a stressor. The concept "pollutant" (in the definition of the NWA) is a subset of the concept "stressor". It should, however, be noted that a stressor may also include a set of variable values that individually would not necessarily have constituted a threat to human or aquatic life, but in combination could pose a threat. For example:

- Substances not in any way necessary for life, e.g. DDT, mercury and cadmium
- Substances necessary in the physiology of life in trace amounts (such as cobalt, zinc and copper) or in moderate amounts (such as salts and acids/alkalis) but which are either present in excess, or, chronically absent.
- Flow which is different (either higher or lower) from that which is natural to the time and place and to which organisms have become adapted over centuries.
- Modification of the in-stream habitat of organisms to a state where it is hostile to the organisms expected at the time and place.
- The presence of biota which are foreign to the time and place and which competes with indigenous biota.
- A critical combination of the first two above, which is manifested as a measurable toxic effect of unidentified origin such as estimated in whole effluent toxicity (WET).

Stressor diversity

Each of these stressors exists because they are deemed a possible cause of a specific effect (e.g. a loss of sustainability). Consequently, any of them could result in "loss of sustainability". The diversity among ecological stressors results from a diversity in:

- Temporal and spatial scale on which stressors have an influence.
- The units in which stressors are quantified.
- The end-points that are applied to the assessment of hazards related to each stressor.

Given that the ultimate guiding principles of water resource quality management are sustainability and equity, there is a need to compare these diverse stressors. The concept of risk is proposed a suitable basis on which stressors can be compared as well as managed.

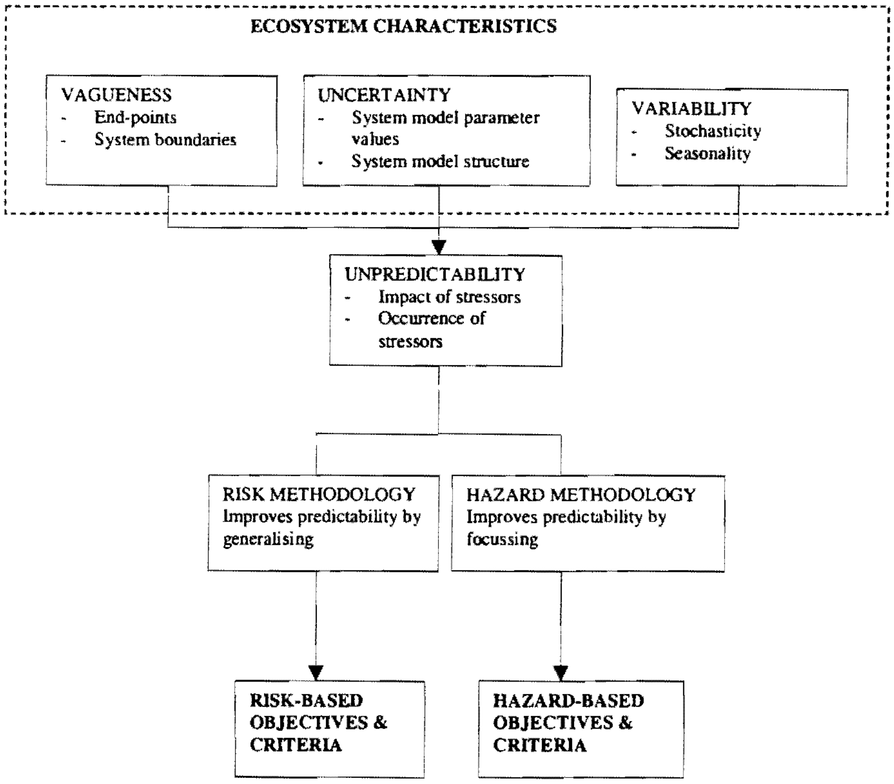

Figure 2
Ecological characteristics and their relationship with risk and hazard methodology

Ecosystem characteristics

A number of biologists consider ecosystems to be unpredictable or even chaotic in its behaviour (Grimm and Uchmanski, 1994). In terms of the NWA goals it is assumed that enough underlying order does exist to draw some conclusions on the response of a system to stimuli and to discount chaotic behaviour. There will still be some unpredictability and these are ascribed to three ecosystem characteristics: variability, uncertainty and vagueness (See Fig. 2).

Variability

Not only is variability commonly encountered, but organisms may be dependent on it. Hydrological conditions, seasonal cycles and variable response thresholds of individual organisms may all contribute to the survival of species. At a deterministic level, this variability may be seen as a source of unpredictability (See Fig.2)

Variability is recognised as a natural characteristic of biota (e.g. Brown, 1993; Grimm and Uchmanski, 1994; Kooijman, 1994). Several types of variability could be encountered. For example, there is a variability in individual response of the biota to a given stressor exposure (e.g. Hathway, 1984). The response variability can be represented by a cumulative response function, which expresses the cumulative fraction of the exposed population displaying a given level of response. This type of function would be analogous to the classic dose-response curve of toxicology, except that the shape of the curve need not necessarily be the same for all stressors. Although these functions may not necessarily be measurable in controlled laboratory experiments, a combination of field observation and expert interpretation is likely to provide an estimate of the stressor-response relationships. In this regard, the use of a Bayesian statistical approach rather than a strict frequentist approach may be indicated (Frey, 1993).

Spatial heterogeneity and stochasticity also impact on many processes in the aquatic environment, such as rainfall and sediment-solute-water interaction, which underlies the variability in the extent to which biota are exposed to stressors (O'Neill et al., 1979; Steinhorst, 1979; Crabtree, et al., 1987; Novotny, et al., 1994; Shine et al., 1995; Canale and Seo, 1996; Kapoor et al., 1997).

In the light of the ubiquity and necessity of variability in the ecosystem, it should not be viewed as a nuisance that can be ignored or even factored out by assumptions. Whichever approach is used in resource management should explicitly recognise this characteristic.

Uncertainty

Uncertainty in the sense used here is a characteristic of the human observer and stems from an imperfect knowledge of the system in point. A comparison between uncertainty and variability is presented in Table 2. Frey (1993) identifies two kinds of uncertainty: model uncertainty and parameter uncertainty.

The model uncertainty in the case of ecosystem models is due to the fact that with imperfect knowledge of a specific ecosystem's processes and mechanisms, there may be several conceptually valid options based on the study of other similar ecosystems or mechanistic models. There may, or may not be some means to weigh the model validity and, hence, the predictions made in this way may all be valid from the point of view of the observer. Only further measurement may reveal which of the models or combinations of models are truly valid. The stress responses may be quite precise, but the discrimination among the model choices

TABLE 2
Some of the characteristics of uncertainty and variability with particular reference to ecological models (based on Frey, 1993 and USEPA, 1997)

Characteristic	Uncertainty	Variability
Source	Lack of empirical knowledge of the observer or imperfect means of observation.	True heterogeneity inherent in a well-characterised population
Impacted by:	Model uncertainty • model structure • range of conceptual models Parameter uncertainty • random error due to imperfect measurement • systematic error (bias) • inherent stochasticity or chaos • lack of empirical basis • unverified correlation among uncertain quantities • expert disagreement on data interpretation	Individualism in response Lack of representative data Aggregation dimension (e.g. time or space)
Description	Probability distribution	Frequency distribution
Effect of more data	Reduces	Same but more precisely known
Applicability of standard statistical data analyses	Understated (due to focus on random error to the exclusion of bias introduced by variability)	Overstated (due to inclusion of measurement error)

may be blurred. This phenomenon is exacerbated by parameter uncertainty. Even when the specific model used to predict effects is known, very often the parameter values are wholly or partially unknown or the number of parameters are unknown. Some sources of parameter uncertainty are listed in Table 2.

These observations imply that in terms of ecologically oriented water resource management, it may be practically impossible to define a specific set of conditions that can be defined as representing "unsustainability". Sustainability will be a function of an uncertain array of possibly stochastic processes. Furthermore, the assessment of sustainability is dependent on a model which is uncertain to a greater or lesser degree and which is subject to variability. The exact point at which the system loses its sustainability can not be described deterministically, but rather in terms of the probability of reaching a condition of unsustainability.

A major problem in ecological goal-driven resource management is the uncertainty in the conceptual model relating the higher level concepts (such as sustainability) to lower level management varaibles (such as quantity and quality). It involves, *inter alia*, uncertainty in stressor-response relationships, uncertainty in the system boundaries and the interactions within the ecosystem (See Appendix 1). Deterministic answers are often not feasible or simply impossible and so decisions have to be based on uncertain information about a variable system. This emphasises the necessity for the use of probabilistic or possibilistic tools in water resource management to ensure protection of aquatic ecosystems.

Vagueness

This is also a characteristic of the human observer, but unlike variability and uncertainty as used above, it is not related to the content of one's knowledge, but to the state or type of one's knowledge. This may result, for example, when different lines of evidence in the assessment of sustainability contribute conflicting information. While this may superficially appear to cast serious doubt on the scientific tenability of the information, this phenomenon may simply result from different levels of assessment (e.g. different spatial and temporal levels, different levels of organisation, etc.). While the solution to this problem is outside the scope of this study, it is clear that a simple determinsistic approach will be inefficient and misleading.

Risk as a concept and an approach

In a colloquial sense, risk may refer to the gravity of the consequences when a mishap occurs or the potential that an undesired outcome may result from an action. The colloquial definition emphasises the hazard (or potential of causing an effect) resulting from an event while the latter definition emphasises the probability. In both cases there is a measure of dimensionality to risk; either the description of the hazard, or the specific consequences for which the probability is estimated.

Definition of risk

The concept of "risk" was defined in 1901 for the actuarial sciences as "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1990, p16) or the probability of observing a specified (undesired) effect as a result of a toxic chemical exposure (Bartell et al, 1992), or, simply, the possibility of suffering harm from a hazard (Haas, 1993). For the purpose of the reserve, a definition is favoured that is essentially dimensionless: Risk is the likelihood that a loss of sustainable ecological function will occur.

This definition emphasises two important aspects:

- An *a priori* decision as to what the undesired event is (i.e. loss of sustainable ecological function)
- A realisation that there is uncertainty about the event which is expressed in terms of a likelihood.

It may not be possible to assess the likelihood of this event directly ('statutory risk') and it may be that the risk of surrogate events may have to be assessed ('surrogate risk') in order to assess the statutory risk.

Hazards and risk

A hazard, in contrast to risk, refers to the potential that a situation has to cause harm. The hazard is not equivalent to the risk it entails. The hazard is a characteristic of the stressor that emphasises what could happen if the ecological entity is exposed to the stressor. It does not express how likely it is to happen since that depends on the situation being assessed.

For example: An endocrine-active substance is discharged to a river. It is known to cause testicular feminisation in fish at a level of 1 mg/ℓ. Its median lethal concentration for fish is about 600 mg/ℓ but its solubility in water is limited to 15 mg/ℓ. At the solubility limit it is unlikely to cause more than 10% mortality in a fish population. There are two hazards involved: mortality and population extinction through inhibition of fertility. If its concentration is managed to just below the solubility limit, the mortality risk is very low, but the population extinction risk is very high. In both cases there may be a hazard of unsustainability, but through different mechanisms. The risk will be determined by, for example, the occurrence of the substance as brief pulses followed by periods of very low concentrations, or, a fairly constant level between 1 and 15 mg/ℓ. It is conceivable that the risk in the first instance is lower than that in the second instance.

Expressions of likelihood

Likelihood is used in the definition of risk because there are sources of uncertainty and variability in both the effect and the exposure components of risk. Likelihood may be expressed in terms of:

- mathematical probability which is a product of probability theory, or
- mathematical possibility which a product of fuzzy logic.

Probability expression of likelihood

For an effect E (e.g. loss of sustainability) the probability that E is true is expressed as P(E). It is customarily assumed that P(E) will have a minimum value of 0 and a maximum value of 1.

P(E) may express either or both of two points of view:

- There is enough evidence to suggest that out of 100 repeated observation of E, in a 100*P(E)% of the observations E will be true, or
- There is enough evidence to make the observer believe that E will be true 100*P(E) % of the time.

The difference in interpretation is that in the first case the emphasis is on the frequency that E is true, while in the second case the emphasis is on the confidence induced by the body of evidence suggesting E to be true.

In many real ecological assessments there are not enough data from which a limiting frequency can be deduced from which P(E) can be inferred. However, there might be enough circumstantial or other indirect evidence that E might be true. P(E) would then express the confidence that E could be true.

Possibility expression of likelihood

A more serious problem than a lack of observations faces the assessment of ecological risk. The effect E might not be a clearly defined event. Loss of sustainability is a case in point. The loss of sustainability (or more precisely the point at which sustainability is lost) is not very clearly defined. This means that it not so easy to define E as being true or not. This calls for a multi-valued logic as opposed to a binary logic to express partial truth such as is found in fuzzy logic (Klir and Yuan, 1995). Possibility theory, which is based on fuzzy logic as opposed to probability theory, which is based on binary logic (Dubois and Prade, 1988) may serve well to express likelihood pertaining to the reserve. Such expression of likelihood in the context of the reserve was investigated by Jooste (2001 a).

Risk and hazard approaches

Resource management implicitly requires predictive ability for decision-making. It would not be sensible to suggest a change in a parameter value unless there is reason to believe that it will result in some advantageous effect.

In predicting or projecting an expected ecological effect there are two major aspects regarding stressors that need to be known: the way in which the target ecological entity reacts to changes in stressor level (i.e. stressor-response) and to what extent the target entity is exposed to the stressor. There are sources of unpredictability in both these aspects.

There are primarily two approaches to deal with ecological predictability problems (Fig. 2): the hazard approach and the risk approach. These approaches are both effect-based, but they differ in the way in which they deal with sources of unpredictability.

The **hazard approach** focuses the basis for decision-making by simplifying both the stressor-response and stressor occurrence by (necessary) assumptions. For example: the response variability, which is an inherent characteristic of the ecosystem, is simplified by selecting a stressor value that corresponds to an assumed "acceptable level of effect". This stressor value is then an assessment criterion value.

The criterion value is then interpreted to mean that all stimulus values less or equal to the criterion are acceptable, while all values above the criterion are unacceptable. The existence of a hazard is evaluated for each stressor value as it occurs.

Consequently, the hazard approach focuses both the stressor-response and -occurrence to single numbers, which are then compared.

The **risk approach** generalises the basis for decision-making

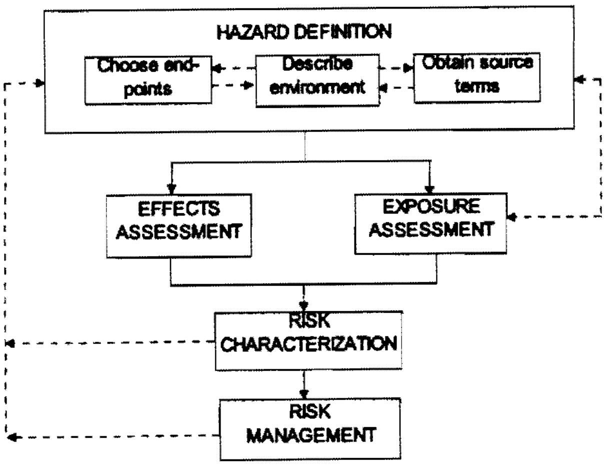

Figure 3
The basic elements of an ecological risk assessment where an ecological stressor and its source has been identified (Suter, 1993)

by incorporating as much of the relevant evidence as possible. It uses as much as is known about the relationship between stressor and response and about the occurrence of the stressor. It recognises that there may be a continuum of response over the stressor value domain at the point or in the area where an assessment is needed.

In the context of the resource management *vis-à-vis* the ecological reserve, where other uses have to be weighed against reserve goals, a risk approach might well be more flexible than a hazard approach.

Ecological risk assessment

Risk assessment is an array of techniques that is primarily concerned with the estimation of the likelihood and magnitudes of events. The likelihood element implies that in principle there is a continuum of risk from infinitely small (practically zero) to very high (practically certain). Due to practical limitations, coarser resolution (e.g. small, moderate, or high) is also used. It has become one of the most widely used techniques in environmental decision-making under uncertainty and has been the subject of intensive investigation by both the USEPA and the American National Research Council (NRC, 1994; USEPA, 1998). Protocols for both environmental and ecological risk assessments have been well-established.

Protocols for the assessment of ecological risk (ERA) have been produced by various organisations such as the USEPA. The basic elements of the ecological risk assessment process are outlined in Fig. 3 and discussed below. A generic adaptation of the USEPA protocol for South African environmental assessment and a more extensive discussion of the elements of an ERA have been produced by Murray and Claassen (1999).

There are a number of features of ERA that need to be considered in applying the methodology in water resource management:

- ERA can be performed at various levels of sophistication depending on the management need and the data input quality. The assessment ranges from qualitative through point estimates to full probabilistic assessments.
- The management goal under the NWA (and, therefore, the statutory end-point) for ERA is loss of sustainability. Assessing the satutory risk is usually difficult since it is unlikely that data will generally be available to assess the likely loss of sustainability in any given stressed aquatic ecosystem. It is more likely that data relating to lower level phenomena are available. A conceptual model (such as the example in **Appendix** 1) is required to project the uncertainty in loss of sustainability from knowledge of the measurable parameters. Such a projection model will relate the surrogate risk to the statutory risk.
- Each stressor risk can be assessed separately and aggregated later. Jooste (2000) and Jooste (2001) investigated a model for aggregating the risk for a number of diverse stressors.
- The ERA process explicitly makes provision for consultation with parties outside the management group. The NWA makes provision for public comment on the reserve. This affords the opportunity to consider a variety of opinions on the reserve. The ERA process also allows for consideration of specific values outside of the scientific opinion inherent in the process.

Discussion

A hazard-based precautionary approach might be administratively ideal. A pragmatic version of a hazard approach was suggested by Van der Merwe and Grobler (1990) by using the pollution prevention approach for hazardous chemicals and the receiving water quality objectives (RWQO) approach for the non-hazardous substances. In terms of the ecological reserve, the distinction between hazardous and non-hazardous is difficult and the aggregation of diverse stressors is not possible with RWQOs. In addition, using hazard-based RWQOs (e.g. those based on the South African Water Quality Guidelines (SAWQG, 1997)) does not allow for effect-based management as implicitly required under the NWA. While the principle of using in-stream objectives is sound, greater benefit would derive from using risk-based objectives (See **Appendix** 2).

The implication of the NWA, stressor diversity and the characteristics of the ecosystem allow for the use of an ecological risk approach because of its formulation in terms of likelihood. In particular, it is noted that:

- The NWA requires sustainable use. This implies that use of the resource needs to be balanced against its protection. A hazard approach to water resource management tends to be inflexible when use is permitted (or even encouraged). This is because only some of the stressor effect information and some of the stressor occurrence information are used to assess resource status. On the other hand, a risk approach allows more of both effect and occurrence data to be used.
- The diversity of stressors that impact on the aquatic ecosystem cannot be handled in an integrated fashion by a hazard approach. Commonly, a hazard will be defined in terms of stressor measuring units such as concentration, flow rate, etc. A hazard approach does not inherently allow for ranking stressors or managing for combined effect. A risk approach has the advantage of placing stressors on a common, practically unitless basis.
- The characteristics of the ecosystem and our knowledge of it such as the necessity of variability and the epistemic uncertainty mitigates against making any information regarding the system and its response to stressors redundant. Such redundancy is

necessarily a part of the hazard approach to resource management. The risk approach, by contrast, tends to be less wasteful of available data.

The use of risk does not preclude a precautionary approach. Precaution is introduced by, for example, conservative assumptions or policies regarding:

- Risk acceptability criteria (what levels of risk are acceptable for each class)
- Acceptability of stressor-effect data (e.g. rejecting data that suggest questionably high tolerance)
- Stressor occurrence estimation (e.g. not accepting stressor degradation for conservative substances)

Although risk assessment may yield continuous assessments, setting risk acceptability criteria could generate dichotomous assessments. Such criteria may comprise of:

- a *de minimis* risk criterion, i.e. a criterion that indicates that the risk is too small to be of any concern and the situation that gives rise to it does not need serious attention, and
- a *de manifestis* risk criterion, i.e. a risk that is unacceptably large and the situation that gives rise to it, one that is unacceptable.

In the present context, where risk is descriptive of a viewpoint of an observer, both *de minimis* and *de manifestis* risk are more likely to be generated in the water resource management policy domain than in a strictly scientific domain. The range between the *de minimis* risk value and the *de manifestis* risk value can be divided into an arbitrary number of values to correspond with the resource classification required under the NWA. These would then give rise to resource risk objectives (RROs).

The RROs would then reflect the aggregate risk of all stressors in the resource (as defined in the definition of the reserve). These RROs could then be used to derive site-specific resource quality objectives that take cognisance of the local surrogate risk parameters as well as the characteristics of the known stressor sources in a catchment. An example of this is given in Jooste (in press).

Conclusions

Ecological risk could serve as a useful approach in certain aspects of water resource management. Interpreting resource classification, as required in the NWA, on a risk base, will assist in deriving resource quality objectives that are both efficacious and flexible.

An ecological risk approach is not a panacea for water resource management. It requires consideration of the scientific data and its relation to human values. It reduces decisions from a purely mechanical process to one that requires explicit action. While this may be difficult in some situations, it increases the flexibility and transparency of the catchment management process while simultaneously assuring that the goal of protection of the ecosystem is attained to the extent possible.

Risk as a tool, although not exclusively dedicated to, is best applied in a risk management framework. In such a framework the objective of risk based decision-making would be to balance the degree of risk to be permitted against the cost of risk reduction (not necessarily only in monetary terms) or against competing risks.

- Formulating a policy for the use of risk-based methods which should serve both to guide the development of an ecological risk assessment ethic in South Africa (e.g. it would address the perception that using risk is merely an excuse for doing nothing (Tal, 1997)).
- Developing a framework for risk-based resource quality management and synthesising this with the current institutional framework.
- Defining and evaluating an acceptable risk range bounded by the *de manifestis* and *de minimis* risks.
- Discretising the acceptable risk range in keeping with the classification of water resources and formulating realistic risk-based objectives in keeping with the ecological reserve.
- Investigating methodologies from the information sciences by which the scarce data and expert knowledge can be brought together to produce the information, particularly the stressor response information, needed to calculate the stressor specific risk.

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Appendix 1
A conceptual model for end-point projection

It is unlikely that data will generally be available to assess the likely loss of sustainability in any given stressed aquatic ecosystem. It is more likely that data relating to lower level phenomena are available. A conceptual model is required to project the uncertainty in loss of sustainability from knowledge of the measurable parameters. A phenomenological inference model for the ecological reserve with a precautionary approach may be based on the following postulates:

- The reference state for the model is the pristine system. The pristine system has all the characteristics (including the potential for sustainable use) that could be wished for. It is assumed that the reference state's only fixed characteristic is its 'degree of correspondence to the pristine state', but that the values of the descriptors used to characterise this state would be spatially and temporally variable.
- For a system that is managed to be under constant stress (as most South African surface water systems are due, to the semi-arid nature of most of the country), integrity (and by implication resilience) is lost more easily than in a comparable system subject to infrequent high intensity stress (Rapport et al., 1995). This means that both acute (in the sense of high-level short-duration) stress, and chronic (in the sense of low-level long-duration) stress should be addressed in resource management.
- It is provisionally assumed that a specific point exists where the sustainability of the system is lost (the system 'crashes' with respect to sustainable use). This point is generally unknown, but the likelihood of approaching this point can be assessed on a "grey scale". The uncertainty in describing this point is similar in the uncertainty in the critical level of loss of integrity that corresponds to this point. The state of integrity of the system is determined by its state of biodiversity and the deviation from the natural temporal and spatial patterns of flow and water chemistry.

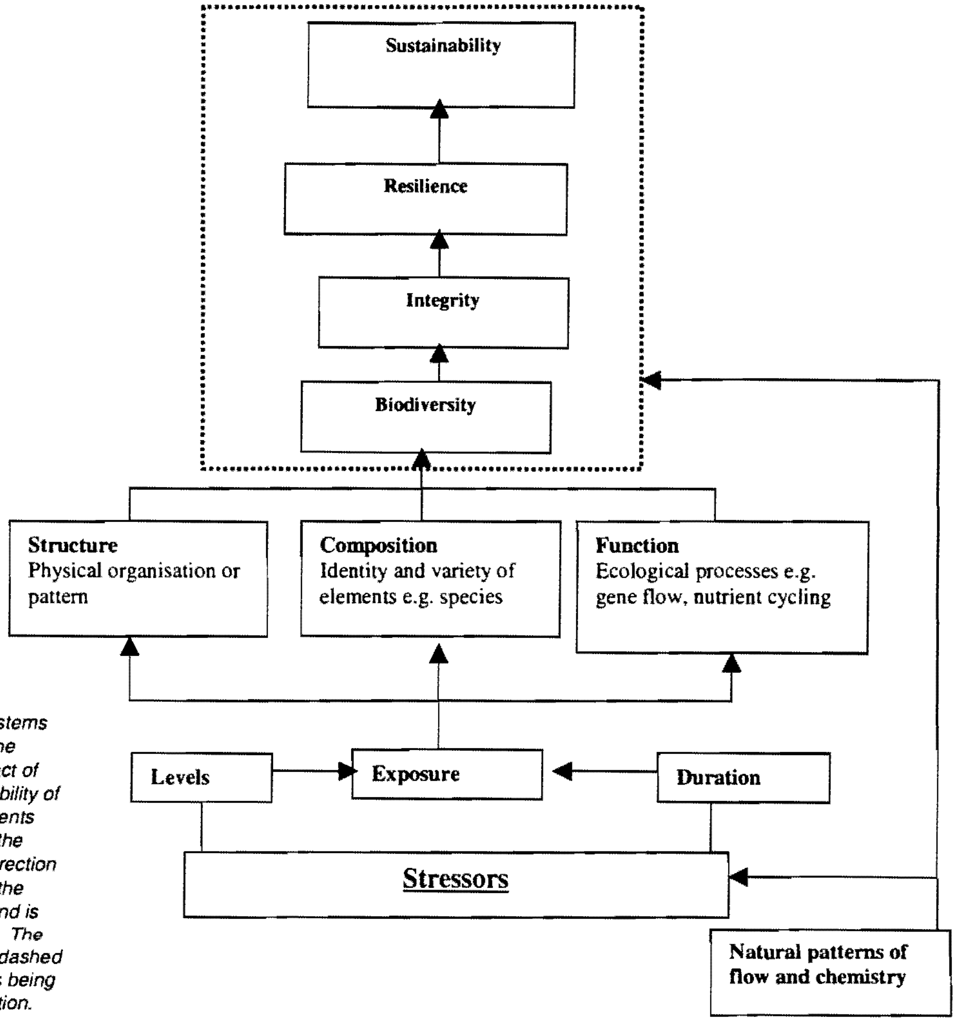

Figure A1
A phenomenological systems model for inferring the uncertainty of the impact of stressors on the sustainability of the system. The elements should be read as : "the uncertainty in...". The direction of the arrows shows the direction of influence and is interpreted as "affects". The elements within the thick dashed line may be combined as being equivalent by assumption.

- Biodiversity is dependent on the composition, structure and function of the system (each at several levels of organization from molecular to landscape level) in relation to what it could have been in an undisturbed, pristine system. Biodiversity as a variable indicating stress is subject to an interpretation of the individual importance of species. Redundancy is possible or even probable in an ecosystem and the real question is how much redundancy could be lost without pushing the system to the edge of some irreversible, catastrophic change (DeLeo and Levin, 1997). The conservative assumption would be that all species are equally important and that loss of species systematically undermines integrity.
- A further precautionary assumption is that the system under consideration is isolated and repopulating from refugiae outside the borders of the system is impossible.

A conceptual phenomenological model based on these postulates is presented in Fig. A1. In this model the arrows indicate how the uncertainty in one variable affects the uncertainty in another. The elements within the thick dashed line are assumed to be logically equivalent in the sense that the epistemological uncertainty in the impact of one on the other is similar. This assumption need of course not hold if more specific information is available.

Each of the propositions regarding impact (represented by the arrows in Fig. A1) of this conceptual model is based on a sense of expectation founded on the assessor's knowledge base, experience and perception of the specific situation being assessed.

Logically, the certainty in a higher level variable cannot be higher than that of a lower level variable. This means that there is a greater uncertainty in the statutory risk than in the surrogate risk. This model helps the assessor to select an end-point and the same time to describe the uncertainty in the risk assessment goal.

Appendix 2
A risk interpretation of the current SAWQG criteria

Suppose a specific effect gives rise to an event E in an ecosystem that is subject to n different stressors. In general, each different stressor i will give rise to E_i. The combined probability of effect is given by (DeFinetti, 1990):

$$P(E)=P\left(\bigcup_{i=1}^{n}E_i\right)=\sum_i P(E_i)-\sum_{i,j}P(E_iE_j)+\sum_{i,j,h}P(E_iE_jE_h) \\ -\ldots\pm P(E_1E_2\ldots E_n) \quad \text{(A1)}$$

where $P(AB)$ denotes the probability of the conjunction of A and B. The form of $P(AB)$ depends on the independence of A and B. In the case where the occurrence of A is logically independent of B, then $P(AB)$ is expressed as $P(A)P(B)$. The resulting boundaries on the effect probability is given by Eqs. (A2).

$$\max_i\{P(E_i)\}<P(E)\leq\sum_i P(E_i) \quad \text{(A2)}$$

A safety factor γ_i where ($\gamma_i \geq 1$) applied to a risk is a_i for stressor i, to accommodate uncertainty of some kind, then the implied risk b_i for stressor i is: $b_i = a_i/\gamma_i$. If the individual stressor risks are assumed to be logically independent, then, from Eq. (A1), the total risk can be expressed as Eq. (A3).

$$P(E)=\sum_i\gamma_i b_i-\sum_{i,j}\gamma_i\gamma_j b_i b_j+\ldots<\sum_j\gamma_i b_i \quad \text{(A3)}$$

Comparing the situations where there are n different stressors present to the one where there are m different stressors:

$$\frac{P(E)_n}{P(E)_m}<\frac{\sum_{i=1}^{n}\gamma_i b_i}{\sum_{i=1}^{m}\gamma_i b_i} \quad \text{(A4)}$$

If $m > n$ then the right-hand side of Eq. (A4) is less than one if γ_i is constant. This implies that if a constant safety factor is used in the derivation of criteria, the total risk to the ecosystem increases as the number of (potentially) additive stressors increase. Alternatively, if a constant total risk is assumed (which should be independent of the number of stressors) then the risk ratio should be 1 and, therefore, Eq. (A4) becomes Eq. (A5):

$$\sum_{i=1}^{m}{}^{m}\gamma_i{}^{m}b_i<\sum_{i=1}^{n}{}^{n}\gamma_i{}^{n}b_i \quad \text{(A5)}$$

If the safety factor is to be independent of the stressor and the individual stressor risk levels are constant then ${}^{m}\gamma > {}^{n}\gamma$, which means that the safety factor is dependent on the number of stressors if the total risk is to kept constant.

In the derivation of the current SAWQG criteria provision is made for a target water quality range (TWQR, abbreviated to T), a chronic effect value (CEV, abbreviated to C) and an acute effect value (AEV, abbreviated to A) (Roux, *et. al.*, 1996; SAWQG, 1997). Although risk is not the explicit basis for derivation, each of these implictly represent a risk a_i, c_i and t_i respectively. By definition $c_i > t_i$, but there is no way of comparing a_i and c_i directly since they refer to different end-points.

There is an implicit maximum total acceptable risk of effect E of $max\{a_i, c_i\}$ for any single substance i. If the management goal is that the substance concentrations are lower than the criterion values, then from Eq.(A2) the total risk, $P(E)$, will be expressed as in Eq (A6).

$$P(E_A)\leq\sum_{i=1}^{n}a_i \\ P(E_C)\leq\sum_{i=1}^{n}c_i \quad \text{(A6)}$$

If all the stressors acted *independently* then, in which case the implicit risk condition is met. However, if stressors k and l, for example, interact with the target organisms by some common mode of action, so that their effect is additive in some way (Calamari and Vighi, 1992), then the probability of their combined effect can be expressed in terms of the joint probability, say $P(E_A/A_k,A_l)$ which, according to Eq. (A3), will always be larger than $max\{a_k, a_l\}$.

This means that if:

- There is any additivity of effect among the stressors present and management up to the criterion levels allowed for each stressor, then the probability of combined effect will be larger than the implied maximum acceptable effect probability. Consequently, management of stressor levels up to the criterion values will logically result in an "unacceptable" level of effect.
- Safety factors had been applied in the derivation of the criteria (Kooijman, 1987), so that the actual risk implied by the criteria is less than the acceptable risk, then the margin of safety afforded by these safety factors depends on the number of stressors assumed to be present (Eq. (A5)). Chapman et al., (1998) point out that current application of safety factors is largely a matter of policy and not of empirical science and that injudicious use may result in useless overprotection.

A model to estimate the total ecological risk in the management of water resources subject to multiple stressors

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Abstract

The disjunctive convolution of independent individual stressor risk is presented as a model to estimate the total expectation of ecological effect for a water resource, subject to several different and metrically disparate stressors. This method makes use of the exposure and effect assessment data of the risk assessment procedure for each individual stressor given that the end-point is the same. A hypothetical case study illustrates how total risk could be used as an ecological goal-oriented tool in catchment management.

Glossary

ERA:	Ecological risk assessment
Hazardous:	Having the potential to cause an (undesired) effect
Stressor:	An anthropogenic substance, form of energy or circumstance that may cause a change in ecosystem integrity
N(x,y) :	The normal (Gaussian) distribution with median x and standard deviation y
LN(x,y):	The log-normal distribution with median x and standard deviation y
Weibull(α, β):	The Weibull distribution with scale parameter and location parameter
$[a, b]$:	The interval from a to b where both a and b are included
(a, b) :	The same interval with both a and b excluded.

Introduction

The management of a water resource with a specific ecological goal in view can be particularly problematic when the water resource is subject to multiple diverse stressors such as chemical substances, deviations from expected flow, habitat degradation etc. An example of this is found in the South African National Water Act (Act 36 of 1998). It makes provision for an ecological Reserve, a quantity and quality of water to (*inter alia*) protect aquatic ecosystems in order to secure ecologically sustainable development and use of the water resource. The provisions of the Act pertain not only to the regulation of discharges to surface water but also to abstraction from the water resource as well as to the quality of the instream and riparian habitat necessary for assuring the protection of the aquatic ecosystem. At the same time, it is recognised that South Africa is a semi-arid country (DWAF, 1986) and consequently a fine balance is needed in water resource management between protection and utilisation. Here the ecological goal of sustainability must be achieved in aquatic ecosystems subject to diverse stressors such as discharge of substances, the abstraction of water and the destruction of the physical habitat which occur to a greater or lesser degree.

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It has been suggested (Jooste and Claassen, submitted to *Water SA*) that a probabilistic effect-based approach has some potential for application to the problem of multiple stressor impacted water resources. A method is suggested whereby an adaptation of the conventional ecological risk assessment methodology can be used to assess the overall risk of multiple stressors in the management of catchments with a view to maintenance of the ecological Reserve.

The problem of a multiple stressor environment

One of the difficulties of ecological water resource management in a multiple stressor environment is the problem of predicting the integrated effect of co-occurring stressors of different types. The disparity among stressor measures necessitates the separate consideration of stressors and their effects. The stressors are then regulated, assessed and controlled separately. At the same time, these stressors may add to a disruptive effect. The integration of effects has been attempted mechanistically on a physiological basis by considering the production of stress proteins (originally referred to as heat shock proteins). These are grouped into three classes:

those related to the heat shock phenomenon;
glucose regulated proteins; and
stressor specific proteins such as metallothionein (Di Giulio et al., 1995; Shugart, 1996).

The stress protein response becomes an integrated signal for environmental stress. While such a mechanistic approach is likely to produce more accurate assessments, its data requirements are extensive. At a more phenomenological level, it may be possible to estimate the probability of stress-induced changes by considering the probability of separate stress events.

Some observations regarding the aquatic ecosystem

The ecological status of a resource is determined by the dynamics and kinetics of interactions of aquatic animals, plants and processes that determine the function, composition and diversity that characterise the ecosystem. Water resource management objectives and their associated criteria must reflect the following inherent ecosystem characteristics if they are to achieve their goal:

A variety of stressors (e.g. habitat, water quality, and flow (Quinn and Hickey, 1994; Armitage and Gunn, 1996; Schofield and Davies, 1996; Dyer et al., 1998)) may be at work at various spatial and temporal scales and yet result in the same unacceptable effect. For example, a fish species may disappear from a river either because of severe chemical contamination, over-harvesting of the species, impairment of crucial breeding habitat or simply because there is no water in the river.

There is an innate and irreducible inter- and intraspecific variability in biotic response to a given stressor. Biotic systems are characterised by variability (O'Niell et al., 1980; Kooijman, 1987; Brown, 1993). The variability observed in the response of organisms may derive from an underlying stochasticity in individual susceptibility (Mancini, 1983; Breck,1988). There is also an underlying stochasticity in aquatic environmental interactions which produces temporal and spatial variability in stressor levels.

There are limits to the scientific certainties about any given natural biotic system which impact, *inter alia*, on the certainty of cause-effect relationships in the particular system. Uncertainty is largely a characteristic of the observer and his deductive processes. Since modelling, whether conceptual or mathematical, often forms a part of the deductive process, uncertainty may derive from:

- uncertainty in future input to the model;
- uncertainty in model structure and parameters; and
- uncertainty in the application and validity range of the model and may well be reducible on presentation of more or better information.

The impact of uncertainty is so severe that the use of quantitative (usually deterministic) predictive models is disparaged by some biologists (e.g. Fryer, 1987). According to Holling (1996), there is "an inherent unknowability, as well as unpredictability, concerning the ecosystems and the societies with which they are linked".

In many natural ecosystems there is a dearth of detailed data about structure, function and composition (e.g. Cairns, 1986; Landers et al., 1988; Munkittrick and McCarty, 1995). Ecological knowledge is often descriptive rather than quantitative. Responses of organisms to stressors are normally continuous and discontinuities are normally an artifact of the resolution of observation. If the test population is large enough or the observation method discerning enough, the response of the population is essentially continuous (e.g. Hewlett and Plackett, 1952; Hathway, 1984)

The above argue strongly for a non-deterministic approach to the impact assessment related to, and management for, ecological goals. Jooste and Claassen (submitted to *Water SA*) suggested the application of ecological risk concepts to resource management in the context of the ecological reserve. The ERA methodology needs to be adapted to assess the overall risk.

Risk assessment

"Risk" has been defined as "the objectified uncertainty regarding the occurrence of an undesired event" (Willet, 1901, *The Economic Theory of Risk and Insurance* quoted by Suter, 1990) or the probability of observing a specified (undesired) effect as a result of a toxic chemical exposure (Bartell et al., 1992). Risk has three necessary components: probability, target and effect; all of which require explicit statement.

"Risk assessment" is an array of techniques that is primarily

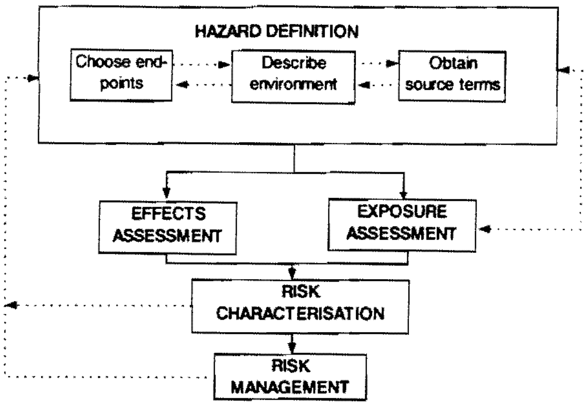

Figure 1
A diagrammatic representation of the predictive use of ecological risk assessment (from Suter, 1993). The dashed lines indicate feedback loops.

concerned with the estimation of the probabilities and magnitudes of events. ERA concerns itself with the estimation of the probability of specific ecological events taking place. These events could comprise a specific effect experienced by a specified target organism (or other ecological entity) when exposed to a stressor. A simplified outline of the procedure is shown in Fig. 1. An important feature is the choice of end-point which implies both target organism (or ecological entity) and level of impact (EPA, 1997a).

The ERA procedure described here is performed at different levels of sophistication (EPA, 1998). The effect assessment is sometimes reduced to generating a number, which, in the estimation of the assessor or the risk manager, represents an acceptable level of effect expressed in terms of a measurement variable such as the concentration of a substance in the water column. This concentration is known under different guises, depending on how it was derived, but is here called the acceptable effect concentration (AEC).

The exposure assessment feature derives a number, which is assumed to represent a suitable exposure scenario (e.g. the worst case exposed organism, reasonable worst case exposure, median exposure etc.), also expressed as a concentration. This is the exposure concentration (EC). Depending on the situation, the EC may either be predicted or measured. In its simplest form, i.e. a screening level risk assessment, the risk characterisation step involves the convolution of the effect level and the exposure level in the form of a ratio. The risk number is calculated as the ratio (DEPA, 1995): R = AEC/ EC. At a screening level, it is only necessary to establish broad categories for this ratio. For example if $R \in [0,1)$ then no further calculation may be necessary; if $R \in [5, \infty)$ then the risk is assumed to be too high and other steps need to be taken to address the situation, while if $R \in [1,5)$ a more detailed risk calculation is needed. At more advanced levels the uncertainty and variability pertaining to the system and its models are brought into the calculation, yielding a probabilistic risk assessment.

The characteristics noted above, of the systems that are to be protected by the implementation of the ecological reserve, make the use of risk-based techniques such as ERA attractive. In an appraisal of the risk assessment and risk management in regulatory programmes, the Commission for Risk Assessment and Risk Management (CRARM, 1996) came to the conclusion "that it was time to modify the traditional approaches to assessing and reducing risks that have relied on a chemical-by-chemical, medium-by-medium, risk-by-risk strategy" and to focus rather on the overall goal of risk reduction and improved health status. They maintain that risk assessment was developed because scientists were required to go beyond scientific observation to answer social questions about what was safe.

Risk convolution

Each stressor acting on an ecosystem produces an individual risk or probability of effect. Each of these individual stressor risks can be estimated by ERA. In order to assess the expectation of all the stressors acting at the same time, the individual stressor ERA outcomes need to be convoluted. There are several mathematical operators that can be used to convolute stressor risk to reflect the total risk, including: maximum, sum and conjunction. In order to explore the use of each of these, it is necessary to formalise the description of the ecological objectives in probabilistic terms.

An ecological objective can be described in terms of events, with an "event" consisting of the information triplet {object, end-point, level}. For example, the information that "more than a 5% decrease in the expected biodiversity may cause an irreversible change in this ecosystem" gives rise to the objective: "the decrease in biodiversity should be less than 5%". This can be encapsulated in the event E = *{biodiversity, decrease, 0.05}*.

The event E can further be partitioned into events (DeFinetti, 1990) that relate to the various types of anthropogenic stress, such as toxicity (t), flow regime disturbances (q) and habitat degradation (h). Therefore, $E = E_t \vee E_q \vee E_h$ where E_t = *{expected number of species, toxic stress effect, 0.05}*, E_q = *{expected number of species, flow regime disruption stress effect, 0.05}* and E_h = *{expected number of species, habitat degradation stress effect, 0.05}*.

The total ecological risk is expressed by $P(E)$, which is the probability of the conjunction of the partitioned events, and therefore:

$$P(E) = P(E_t \vee E_q \vee E_h) \qquad (1)$$

As a general case, suppose an event E involves a specific level of effect (specified by the assessor or risk manager) in an ecosystem subject to n different stressors. Therefore, each stressor i will give rise to E_i. The combined probability of effect (in set theoretical terms) is given by (DeFinetti, 1990):

$$P(E) = P\left(\bigcup_{i=1}^{n} E_i\right) = \sum_i P(E_i) - \sum_{i,j} P(E_iE_j) + \sum_{i,j,h} P(E_iE_jE_h) - \ldots \pm P(E_1E_2 \ldots E_n) \qquad (2)$$

If E_t, E_q and E_h are all logically independent, then probability of the conjunction of individual ecological effects reduces to the product of the individual effect probabilities, and hence the application of Eq. (2) to Eq. (1) yields Eq. (3):

$$P(E) = P(E_t) + P(E_q) + P(E_h) - [P(E_t)P(E_q) + P(E_t)P(E_h) + P(E_q)P(E_h)] + [P(E_t)P(E_q)P(E_h)] \qquad [3]$$

It is recognised that $P(E_t)$, $P(E_q)$ and $P(E_h)$ are joint probabilities of effect ε_x and exposure x so that: $P(E_x) = P(\varepsilon_x, x) = P(\varepsilon_x | x)P(x)$, where $x \in \{t, q, h\}$.

A distinction is made between logical dependence and causal dependence (Jaynes, 1996). Two events A and B are logically dependent if, for example, the occurrence of A implies the occurrence of B. This is different from the proposition "A causes B". If a reduction in biodiversity due to toxicity is inferred from the information at hand, then there is no possibility of inferring that reduction of biodiversity due to habitat stress will occur. This should not be confused with the situation where, for example, data at hand indicate that the probability of mortality due to toxic stress in conjunction with habitat stress is greater than that predicted by Eqs. (2) or (3). $P(E_x)$ should not be confused with $P(\varepsilon_x)$ (see below).

$P(\varepsilon_x | x)$ is defined as the probability of an effect given the event that stressor X is present at level x. This information is derived from a probabilistic stressor response relationship, which predicts the probability of a specified effect (of the same type as in the original n-tuple definition; i.e. the expected number of species in this case) as a function of exposure to a stressor. This implies that the value of $P(E_x)$ can simply be estimated from a probabilistic stressor response relationship and the probability of occurrence of exposure to a stressor x. Stressor response relationships are often evaluated empirically, although it might be necessary to partition each of the events in Eq. (1) into component events in order to get to a level at which sufficient empirical data can be collected to evaluate the event probability.

Furthermore, the effects ε_x may not be functions of one stressor only. It may be necessary to partition the event "existence of stressor X" into events that signify the occurrence of stressors that collectively manifest as stressor X: i.e. X is partitioned into occurrence of stressors (X_1, X_2, ...X_n), where there are n stressors that make up the class of stressor X. Due to interactions among stressors, it may be necessary to evaluate $P(\varepsilon_x | X)$ where all n different stressors are present at the same time. Most often this will not be possible experimentally (except perhaps in the case of toxic stress), so that simplifying assumptions will have to be made. However if events X_i are logically independent then this reduces to (DeFinetti, 1990):

$$P(\varepsilon_x | X) = \sum_j (PX_j) \cdot P(\varepsilon_x | X_j) \qquad (4)$$

It might be, that although the stressor occurrences X_i and X_j are independent, the effect ε is dependent on the co-occurrence of X_i and X_j. This might be due to some mechanistic interdependence such as synergism or antagonism in which case the occurrence of (X_i, X_j) might manifest as a new stressor Y. In this case $P(\varepsilon | X_iX_j)$ would be given by $P(\varepsilon_Y | Y) = P(\varepsilon, Y) | P(Y)$. Therefore, $P(\varepsilon, X_i, X_j) = P(X_i)P(X_j)P(\varepsilon | Y)$, where the value for $P(\varepsilon | Y)$ has to be evaluated experimentally. However, cases of true synergism among toxics, for example, are reported to be rare (Calamari and Vighi, 1992). The occurrence of synergism among other stressors may be possible.

A hypothetical case study

In an ERA for a stretch of river it was agreed between the risk manager and the risk assessor that the sustainability of the aquatic ecosystem can be expressed in terms of the end-point "a 5% decrease in biodiversity". Furthermore, three sources of stress (i.e. the hazards) were isolated:

Stressor 1 is the modification of the streambed and riparian zone resulting in destruction of habitat (independent of flow). This is reflected in habitat degradation which is expressed (hypothetically) as a percentage, where zero indicates no degradation and 100 denotes complete degradation. In the assessment, it is found that there are practically pristine sections as well as degraded areas in the river reach, so that the habitat degradation can be described by a normal distribution (see Table 1). It is proposed that the response of the system to habitat degradation (all else being equal) can be described by a Weibull distribution (Fig. 2a).

TABLE 1
Stressor Magnitude and System Response Modelling Functions

Stressor	Stressor response function P(E|x)	Stressor magnitude distribution P(x)
Habitat	Weibull(5, 50)	N(25, 7)
Flow	1-Weibull(15, 7)	LN(12, 1.3)
Toxics (Scenario 1)	Weibull(3, 2.715)	LN(3.8, 1.25)
Toxics (Scenario 2)	Weibull(3, 2.715)	LN(1.9, 1.25)
Toxics (Scenario 3)	Weibull(3, 2.715)	LN(0.95, 1.25)
Toxics (Scenario 4)	Weibull(3, 2.715)	LN(0.475, 1.25)

Stressor 2 is the water depth in the river. This is assumed to be directly proportional to the flow which is log-normally distributed for the reach under investigation. It is accepted by the river ecologists on the risk assessment team that the response of the system to this measure can be described by an adapted Weibull function as shown in Table 1 and Fig. 2b.

Stressor 3 is the presence of toxic substances in the river. These substances are unidentified and were established by whole effluent toxicity testing at the source discharge to the river. The level of these substances is expressed in terms of toxic units. For this situation a toxic unit has been defined as: 100/LC5, where LC5 is the 5th percentile of the mortality distribution for the test organisms with the concentration expressed as a percentage (DEPA, 1995). The toxic units were found to be log-normally distributed. From ecotoxicological studies, the system response to these toxics is approximated by a Weibull function (Fig. 2c).

It is assumed that the flow regime as described will not result in further habitat degradation by inducing changes in channel morphology. There has been no evidence to suggest an interdependency among the stressor effects. Consequently, the occurrence of effects resulting from these stressors is logically independent by default assumption.

Total risk calculation

The convolution expressed in Eq. (3) was used. The stressor-response profile is expressed as the probability of "a significant ecological effect" in the river reach and the result is expressed as the cumulative probability of effect ($P(\varepsilon_x | X)$. This type of result may be obtained from a site-specific study, expert opinion or system simulation modelling.

The stressor-specific probability of effect is calculated from the product of the stressor probability density and the probability of effect to give the probability density of effect for this river reach for each stressor X (stressor risk $p(E_x)$).

Since these stressors have been assumed to occur independently, Eqs. (3) and (4) were solved iteratively by randomly selecting the stressor risks from their respective density profiles to obtain the risk distribution for these specific conditions in this river reach. The random stressor magnitudes were calculated as described in Frey and Rhodes (1999). One thousand random samples were selected for each stressor. The stressor profiles, and conditional response probabilities are shown in Figs. 2a, b and c. The calculated risk distributions are shown in Fig. 3.

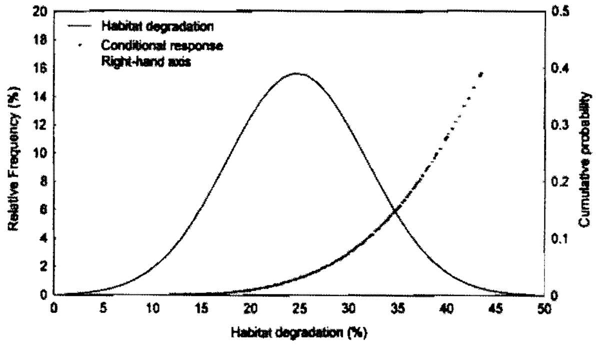

Figure 2a
Habitat degradation distribution as used in the Monte Carlo simulation and the conditional probability of system response (points referring to the right-hand ordinate scale)

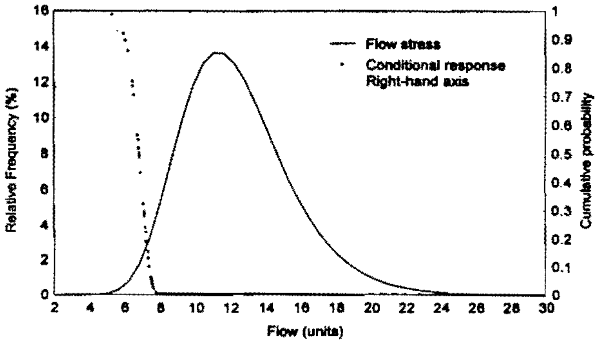

Figure 2b
The flow-related stressor magnitude distribution (solid line) and the corresponding conditional system response probability (point referring to the right-hand ordinate scale)

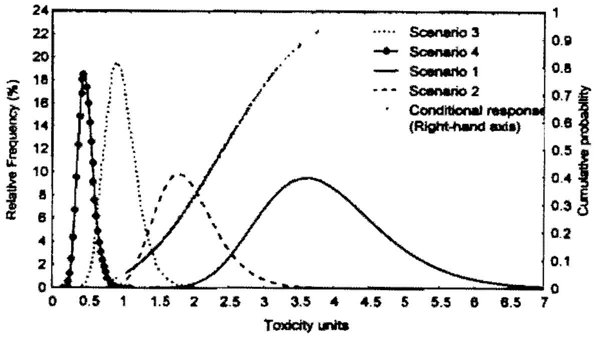

Figure 2c
The toxic unit distribution for the four scenarios described in the text (lines referring to the left-hand ordinate scale) and the conditional system response profile for the toxic substances (the points referring to the right-hand ordinate scale)

Risk ranking

The contribution of each stressor to the risk expectation for a river reach may vary depending on the stressor-response profile and stressor-probability profile. The conjunctive convolution model (Eq. (2)) predicts that, depending on the risk level allowed, differ-

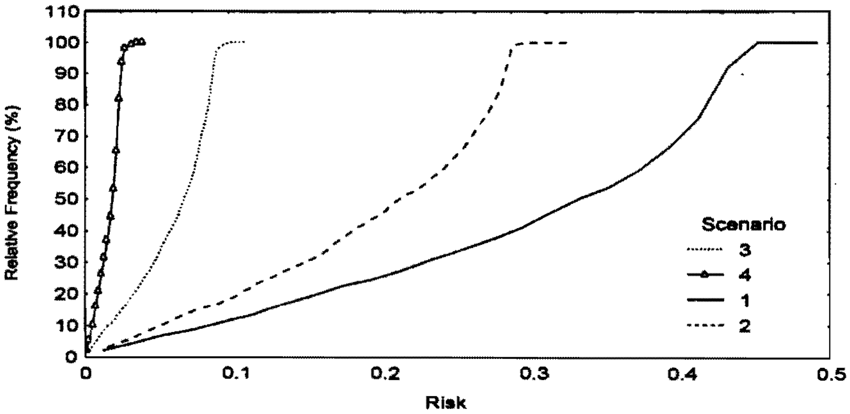

Figure 3
The cumulative probability profiles of the overall risk from the Monte Carlo simulation of the scenarios described in the text

ent stressors could dominate the overall risk in a catchment. It is possible to rank the risks, rather than the hazards, in a catchment and focus on those. In the example above, it can be seen from the stressor profiles, that the presence of toxics appears to dominate the risk contributions. The management objectives for stressors giving rise to lower risks could be set at levels in some way representative of the lower risks (e.g. median lower risk, i.e. median stressor risk excluding the dominant stressor risk). The sub-dominant stressors in the catchment need only be monitored (e.g. by means of the stressor probability profile) until the dominant stressor had been addressed. Periodic recalculation of stressor risks will reveal either the appearance of a new dominant stressor or the overall acceptability of the integrated risk.

The ratio of the individual stressor risks to the total risk is depicted in Fig. 4. It is apparent that in Scenario 1 (Table 1) above, the toxicity in the river is the major contributor to overall risk.

This can also be seen by inspecting the position of the response curve in relation to the stressor magnitude profile in Fig. 2c. Based on this assessment, it would seem likely that the relatively high overall risk (90th percentile of about 0.44) can be ameliorated by managing the system to a lower toxic unit level. For Scenario 2, the toxic unit median is set to 1.9. The corresponding overall risk 90th percentile is now less than 0.3 but still too high. For Scenario 3, the toxic unit median is adjusted to 0.95 and for Scenario 4 the toxic unit median is adjusted to 0.475. The individual risk ratio's for Scenario 4 is shown in Fig. 5.

A comparison of Figs. 4 and 5 shows that the habitat-related risk has become more significant even though it is still less that the toxic substances risk. The overall (total) risk in the river is now at a more acceptable level (Fig. 3), but it is clear that a point will be reached where the overall risk can no longer be reduced by simply managing for the most apparent stressor, i.e. the toxic substances in the river.

It has been recommended that uncertainty and variability be separated to provide greater accountability and transparency in a probabilistic assessment (Frey, 1993; EPA, 1997b). A two-dimensional Monte Carlo simulation with bootstrap sampling was performed in order to assess the impact of uncertainty in the stressor-response relationships on the 50th and 90th percentiles of the risk distribution. For the hypothetical case under discussion, it was assumed that one of the major problems in setting up a stressor-response relationship would be to establish where the no-effect (or more precisely, the undetectable effect) and unacceptable-effect levels would be. For the sake of illustration, assume that the location parameter (β) of the Weibull function would have the greatest uncertainty and that the uncertainty in β can be described by a normal distribution. The increase in uncertainty is reflected in an increase in the relative standard deviation (RSD, ratio of

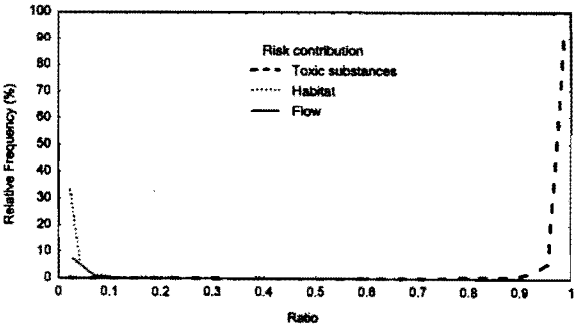

Figure 4
The ratio of stressor-specific risk to the overall risk for Scenario 1

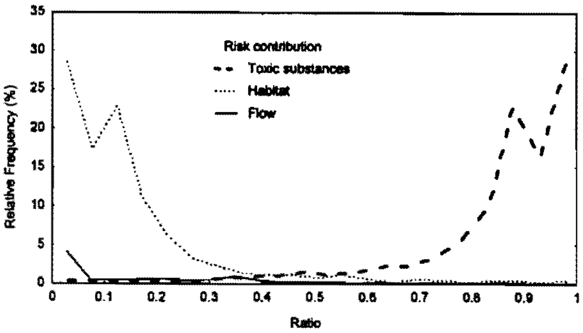

Figure 5
The ratio of stressor-specific risk to the overall risk for Scenario 4

standard deviation to median) of this uncertainty distribution. RSD values of 0.05, 0.1, 0.15 and 0.2 were used. The parameter values of Scenario 1 were used for comparative purposes. One hundred bootstrap samples from this distribution were drawn. Frey and Rhodes (1999) showed that a non-parametric method could be used in this case to select percentiles. The 50th and 95th percentiles of the overall risk distribution were established by ordering the risk values generated from 1 000 random stressor value samples and by selecting the 500th and 950th values.

From Figs. 6a and b, it is clear that there is a significant probability that the overall risk can be underestimated when there is uncertainty in the stressor-response parameters. This would, however, be dependent on the form of the stressor-response function as well as on the uncertainty distribution.

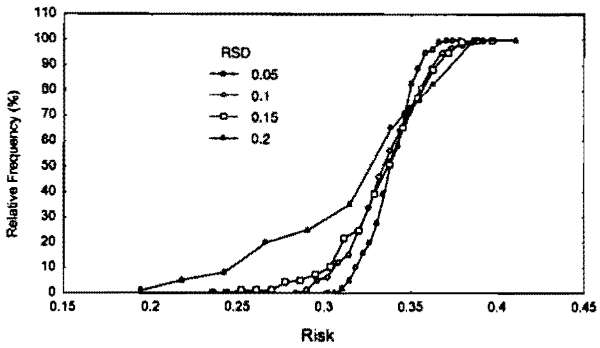

Figure 6a
The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the median risk value

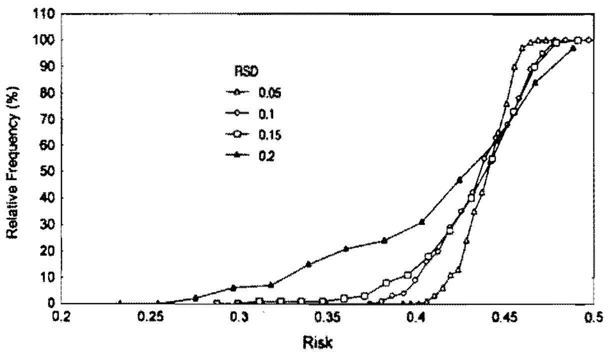

Figure 6b
The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the 95th percentile risk value.

Discussion

The left-hand side of Eq. (1) may, for example, represent the total allowable risk for a specific class of river which, in the case of the ecological reserve, may be determined by the river classification. The implication of the right-hand side of Eq. (3) is that if the individual stressor risks are defined and quantifiable, these can be managed by "trading-off" risks among stressors (as shown in the scenario exercise above) and therefore also among stressor sources. Further reduction of the risk may, for example, be effected not only by reducing the toxics concentration but also by reducing the habitat degradation. In principle, this greatly extends the management possibilities, although in practise there would likely be some bounds on the extent to which trade-offs can be accommodated, the reason being that the probabilistic approach followed here is phenomenological rather than mechanistic. Consequently, the focus is more on the expectation of an effect than on the mechanisms that caused the effect. At stressor levels representing high risk it becomes more critical that the stressor response relationships be well characterised due to the influence non-linearity may have on the expected stressor effect. At lower risk levels, it may well be possible to accommodate a trade-off among stressors. This could be particularly important when stressor discharge rates in a multiple discharge environment are being optimised to economic or technological constraints.

The evaluation of the terms in Eqs. (3) and (4) has been glossed over. In a highly standardised effect-scenario-driven ERA, such as that used in the European Union (Van Leeuwen, 1997), the estimate of stressor-probability profile, $P(x)$, may bear the greatest uncertainty. However, the stressor-response projection may have an equal, if not larger, impact on the overall uncertainty. The discipline of ecotoxicology needs to be used extensively to evaluate the response probability of toxics. Furthermore, the assumption of water depth as a stressor is far too simplistic to be of real value but it was used simply by way of illustration. It seems more likely that deviation from expected virgin run-off may be a stressor. However, much work is being done from which flow-related stress and flow-related stressor-response information can be drawn (e.g. King and Louw, 1998; Hughes and Münster, 1999) and some experimental and or observational data exist from which the possibility of effect can be inferred (e.g. Chessman et al., 1987; Quinn et al., 1992; Cooper, 1993; Roux and Thirion, 1993; Thirion, 1993). It appears that much more research is needed to assess effects at *ecosystem* level. Effect data for toxic substances exist mostly at the *individual organism* level and, to a lesser extent, at the population level, while effect data for the other stressors exist largely t the population and community level. However, more realistic risk assessment is still hampered by a lack of knowledge of conditional probability of effect at higher levels of organisation. As a simplification, it is often assumed that an impact at the lower level of organisation (where the data exist) necessarily implies an impact at the higher level of organisation. Consequently, the risk predicted at the lower level of organisation is at least as great as that predicted at the higher level of organisation since the probability of a logical consequent cannot be greater than that of the antecedent. Although this is a reasonable starting point, if all the interactions have not been accounted for and the conditional probabilities evaluated, this assumption could be seriously in error. As a result, the calculation above, and indeed any risk assessment based on such a premise, could be seriously in error.

Probability as an epistemic issue

Interpretation of the terms "risk" and "probability" has a fundamental impact on the approach to, and application of, risk methodology (Power and Adams, 1997; Suter and Efroymson, 1997). The interpretation of probability is crucial to decision-making in data-poor ecological management situations. The "frequentist" approach (Jaynes, 1996), sees probability as the limiting frequency of an occurrence over a large number of observations.

In contrast, probability can be seen as a subjective expression (not necessarily dependent on repetitive observations) needed to project from the domain of uncertainty by the means of prevision to the domain of certainty. "Prevision, ... consists in considering, after careful reflection, all the possible alternatives, in order to distribute among them, in the way which will appear most appropriate, one's own expectations, one's own sensations of probability" (DeFinetti, 1990). With this view in mind, probability, and by association risk, could be seen as epistemic of the specific combination of situation and assessor.

Regulatory decision-making in the field of ecology is largely dependent on a descriptive conceptual knowledge of ecosystems, often only supported by patchy observation. Observations of multiple replicates of experiments are often not available or simply impossible. What often needs to be considered is the expert prevision pertaining to a specific situation. Predictive ecological risk is essentially an expectation of an effect, a prevision based on best available knowledge of the assessor's knowledge of and expertise in dealing with, what are as yet, unobserved events in a complex system. The calculated ecological risk values are therefore an expression of the assessor's expectation, taking into consideration the scientific information at hand.

Possibility theory (based on fuzzy set theory) (DuBois and Prade, 1988) may be better suited to the kind of situation where semi-quantitative expert opinion, such as in ecology, is the basis of the decision-making process. A fuzzy mathematical approach to ecological risk has been used (e.g. Ferson and Kuhn, 1992; Ferson, 1994) and possibility theory merits investigation as a total risk estimation tool.

Conclusion

Modelling the total ecological risk as the disjunction of independent individual stressor risks can be applied to the management of a water resource subject to diverse stressors. A risk-based approach (as compared to a hazard-based approach) affords greater flexibility to the management of diverse stressor sources by maintaining a common basis for comparing the various stressors and thus creating the opportunity of prioritising and "trading" among stressor scenarios. At the same time the overall risk can be related to management classification of a water resource, providing a basis for developing class-related stressor criteria on a site-specific basis.

It is a truism that the quality of the predicted risk can be no better than that allowed by the information on which it was based. Clearly, research invested into improvement of both the ecosystem inference models and the mechanistic stressor-response and stressor-prediction models will improve the resource management flexibility.

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A possibilistic approach to diverse-stressor aquatic ecological risk estimation

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Abstract

A possibilistic approach to assess the risk of co-occurring stressors in an aquatic ecosystem based on the use of fuzzy sets is illustrated at the hand of a hypothetical case study. There are two aspects of importance: a fuzzy stressor response relationship where the response may have reference to a lower level end-point, and a rule-based inference model relating the occurrence of low-level stressors to a high-level ecological goal such as sustainability. The stressor-response is expressed as a conditional possibility. The possibility and necessity measures of the disjunctive composition of the stressor-response with the possibility distribution of the stressors yield an estimate of the ecological risk. Such a possibilistic approach may well serve as a screening procedure in multiple stressor resource management when only qualitative risk assessments are needed.

Introduction

The South African National Water Act places a premium on water supply for basic human needs and for the sustainable development and use of the aquatic ecosystem. This is reflected in the reserve. The ecological component of the reserve has been defined as that level of quantity and quality necessary to ensure the sustainable development of the water resource (NWA, 1998). The ecological reserve is a water resource management instrument for aquatic ecosystem protection to ensure sustainability in the use and development of the water resource. As a practical management measure, the capacity of the water resource to maintain its sustainability can be discretised into different management classes (MacKay, 1998) corresponding to different levels of risk that the resource may lose its sustainability.

Risk is used here in the sense of the likelihood that a specific undesired event would occur. This likelihood may be expressed in terms of either probability or possibility. In probabilistic risk assessment, it is assumed that this event is crisply defined, i.e. it is possible to decide whether the event has occurred or not. However, the nature and epistemology of the event would determine how likelihood is expressed. Possibility theory offers the option of addressing fuzzy events where the event is perhaps epistemologically vague.

A point of departure in this paper is the recognition that in assessing the risk of the aquatic ecosystem losing its sustainability:

- there are several stressors (such as chemical substances, flow reduction and habitat degradation) that may be present simultaneously and that may result in responses such as loss of sustainability (although the mechanics of these impacts may differ), and
- unambiguous quantitative and possibly even quantitative site specific data may often be lacking.

An argument will be presented for the application of a fuzzy approach to aquatic ecological risk. Two types of ecological risk may be defined depending on how the likelihood measure is expressed: a risk based on a possibility measure (referred to as "ecological concern") and a risk based on a necessity measure (related to the possibility measure and referred to as "ecological dread"). These are illustrated by a hypothetical application to water resource classification.

Rationale for a fuzzy approach

The term "sustainability" is not defined in the NWA. For the purpose of discussion, it is assumed that ecological sustainability refers to the ability of a system to maintain an acceptable level integrity subject to anthropogenic stress. Concepts such as sustainability and integrity may be spatially and temporally scale-dependent and the knowledge of the mechanisms underpinning these phenomena is vague (Costanza et al., 1993, De Leo and Levin, 1997). Variability is both a normal and sometimes a necessary ecosystem characteristic to certain ecosystem processes. "Therefore, in managing ecosystems, the goal should not be to eliminate all forms of disturbance, but rather to maintain processes within limits or ranges of variation that may be considered natural, historic or acceptable" (De Leo and Levin, 1997).

Not only must natural variability be accounted for in the management process, but also uncertainty and, in some cases, vagueness. Definitions of ecosystem integrity varies; e.g. "the maintenance of the community structure and function characteristic of a particular locale or deemed satisfactory to society" (Cairns, 1977) or "the capability of supporting and maintaining a balanced, integrative, adaptive, community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley, 1981). Terms such as "deemed satisfactory"; "balanced", "comparable" and "natural" in these definitions are, without further qualification, essentially vague and subjective. This means that in terms of the risk assessment under the NWA, the end-point is vague.

In addition, the system boundaries, the response to stressors and the stressors themselves may only be known qualitatively. The functional entities that best reflect the goals of ecosystem management may only be vaguely identifiable. Consequently, in dealing with ecological risk in the context of protective ecosystem management, it would be advantageous to use a paradigm that is

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adapted to address both uncertainty and vagueness. This could be accomplished by using the framework of possibility theory (as opposed to probability theory), which is based on the use of fuzzy logic (as opposed to 'crisp' logic).

Probabilistic vs. fuzzy risk

Risk is a way of expressing the uncertainty of observing some event (Suter, 1993). The use of risk techniques in decision-making is largely motivated by the variability and uncertainty observed in dealing with ecosystems and has been used extensively in a number of countries (e.g. USEPA, 1996; Pederson, et al., 1995). Probabilistic risk assessment depends crucially on the ability to derive some expression of probability for a stressor variable. Conventionally, imperfect information has been dealt with either by probability or by interval analysis.

Probability theory has, over a period of 200 years, developed a calculus to deal with stochasticity. A problem with probability theory in ecological risk assessment may relate to the interpretation of what is really represented by probability (Dubois and Prade, 1988). The frequentist approach sees probability as the *limiting frequency* of observed, *clearly defined* events. The first major obstacle in assigning probability distributions for ecological variables is the lack of enough system-specific information to estimate these limiting frequencies. The alternative Bayesian approach circumvents the frequentist dilemma by using probability as a descriptor of the state of knowledge about an event or proposition (Jaynes, 1996) and is often much better suited to generating the necessary distribution data.

The second (and possibly more critical) problem facing ecological risk assessment and risk management is the difficulty in defining the system uniquely at an operational level. The boundaries of ecosystems, communities and even populations, for example, are notoriously vague. This complicates the use of both frequentist and Bayesian statistics, which deal with such vagueness with difficulty. Mathematically, this vagueness, superimposed on the complexity of ecosystems, the elements of which may exhibit stochastic behaviour, results in analyses that become intractable to conventional mathematics. The resulting ecosystem models exist largely as lexical system descriptions. In analyzing a complex multidimensional system, a state could be reached where, even if uncertainty and variability could be quantified, the results would be difficult to interpret (Dubois and Prade, 1988). As the complexity of the system or model of a system increases, a point could be reached where "our ability to make precise and yet significant statements about its behaviour diminishes until a threshold is reached beyond which precision (or relevance) becomes almost mutually exclusive characteristics" (Zadeh, 1973)

Working with incomplete data, ecologists may have to deal largely with judgement, which by its nature has at least an element (if not consisting entirely) of subjective opinion. Possibility theory in contrast to probability theory, "offers a model for the quantification of judgement which allows a canonical generalisation of interval analysis" (Dubois and Prade, 1988) which has been used in the analysis of uncertainty in the physical sciences.

Risk estimation in ecosystems has been shown to be influenced by both uncertainty and variability (e.g. Frey, 1993, Frey and Rhodes, 1999), which argues for a probabilistic rather than a deterministic approach in assessment. The concept of risk contains the elements of:

- value ("what is being threatened"),
- extent ("how badly"),
- the likelihood of a) and b), and
- assessment ("what does it mean").

Applying possibility theory to assessment of ecological reserve-related risk

For discrete events ω with a possibility distribution $\pi(\omega)$, the possibility measure $Poss(A)$ and the necessity measure $Nec(A)$ are defined by Eq.1.

$$\begin{aligned} Poss(A) &= \sup\{\pi(\omega) \mid \omega \in A\} \\ Nec(A) &= \inf\{1-\pi(\omega) \mid \omega \notin A\} \end{aligned} \tag{1}$$

Some of the differences between probability measures and possibility or necessity measures are:

- The *probability* of the sure event is assigned the value 1. For a number of events, the cumulative probability of all possible events is assigned the value 1. A *possibility* of 1, however, does not imply that the event is sure, only that it is entirely possible.
- The knowledge of the probability of an event completely determines the knowledge of the contrary event. Knowledge of the possibility or necessity of an event is less strongly linked to the knowledge of the contrary event. To establish the certainty of an event, it is necessary to know both the possibility and the necessity of the event.
- Probability deals with precise but differentiated items of information. Possibility reflects imprecise but coherent items.
- A central requirement in probability theory is the additivity of the probability of independent, mutually exclusive (disjoint) events. This requirement, generally, does not hold for fuzzy likelihood measures.

These characteristics of possibility theory make possibility measures well-suited to reasoning in an uncertain environment where it is often desirable not to set the relationship between the evidence one has for an event (degree of necessity) and the evidence that weighs against it (1-degree of possibility) too rigidly. In addition, it might be prudent to consider whether one's knowledge that an event (such as loss of sustainability) might occur, also defines the possibility that the event might not occur. In other words, does one's *knowledge of the ecosystem* allow for the law of the excluded middle of Aristotelian ('crisp') logic?

Variability: an inherent and practically irreducible characteristic of a biotic system, stemming from the innate stochasticity underlying processes in the ecosystem.
Uncertainty: epistemic of the observer stemming from imperfect information, due to limitations in observation, modelling or interpretation of system-related data, for example.
Vagueness (or fuzziness): a lack of clarity in the definition of the set of values attached to the object.
Ambiguity: largely associated with language, where the definition of the object is vague or refers to several different reference sets simultaneously.

The regulatory end-point E

In an ecological risk assessment implicit in the classification in terms of the reserve, the "regulatory" undesired event, E, is defined by the NWA as "loss of sustainability". This is a fuzzy event in the light of the foregoing. The management classes in the NWA correspond to differences in the likelihood of this fuzzy event occurring.

This definition of E implies that it is a dichotomous characteristic of the system; anything less than full sustainability means unsustainability. It does not mean that important related characteristics such as resilience and integrity need to be dichotomous as well. There might be levels of resilience and integrity less than 100% that still result in sustainability. E may be epistemologically vague, in that the knowledge of what constitutes E (or $\neg E$ i.e. "not E") may be imperfect. An assessment of the "likelihood of E" may be a reflection of the epistemology of the values of the parameters defining the critical point defining E. Consequently, the evidence one has that a certain set of parameter values corresponds to E and the evidence that it corresponds to $\neg E$ might not be complementary in the sense that one's knowledge of E occurring does not define one's knowledge of E not occurring. There might, therefore, be a set of parameter values for which it is not possible to make a clear assessment of either the likelihood of E or the likelihood of $\neg E$. The "likelihood of E" is interpreted as the degree to which the observed situation corresponds to E.

Ecological concern and dread

The likelihood aspect of risk can be expressed in terms of possibility theoretical concepts. $Poss\ (E)$ could be used to express the possibility that effect E would occur. This does not carry the same weight as the probability of E, P(E). It is always true that $Nec(E) \leq P(E) \leq Poss\ (E)$. This means that Poss ($E$) expresses an epistemic possibility that E could occur and therefore, $Poss\ (E)$ expresses a weaker claim than P(E). More appropriately, $Poss\ (E)$ might designate the degree of "ecological concern".

On the other hand, $Nec(E)$ expresses the cumulative evidence of the necessity that E must occur. This is a much stronger claim that P(E) and may appropriately be expressed as the degree of "ecological dread". Both ecological concern and ecological dread express the accumulated evidence about the likelihood that the undesired event E will occur.

There are three aspects to the assessment of ecological risk in the aquatic environment that are important in the context of the reserve:

- The estimation of the aggregate likelihood of $Poss(E)$ or $Nec(E)$ when diverse stressors occur together,
- The confidence in $Poss\ (E)$ or $Nec(E)$ on projecting E from other available data and
- The formulation of the relationship between $Poss\ (E)$ or $Nec(E)$ and the stressor value.

Aggregating diverse stressors

There are a number of different stressors that could result in loss of sustainability. Assume, for example, that flow deficiency (i.e. degree to which the flow is less than that expected in the natural hydrological cycle), toxic substances and habitat degradation are typical stressors in a system being assessed. In order for E to occur, it is assumed that:

- An environmental variable X with value x, only becomes a stressor if it can result in E, i.e. in the present context, stress is not defined if a variable is within its natural range of variability. Furthermore, there exists a critical value x_0 at which E occurs. Our knowledge (rather than the inherent nature) of E as well as x_0 make both fuzzy quantities. The likelihood of E occurring (both $Poss\ (E)$ or $Nec(E)$) is a function of x. The stress E_X, which is used here in the sense of the extent of the effect E being produced as a result of stressor X, depends on a fuzzy causal relationship $E|X$ and an occurrence of stressor X, where the X is a fuzzy set of stressor values which correspond to x_0 and which is defined in terms of the degree to which a value x corresponds to $x_0 : X = \{x |\ \mu_X(x) = \pi(x{=}x_0)\}$.
- Any of the stressors could result in E, irrespective of whether they occur alone or together. The *ecological concern* would refer to the *possibility* that *any* of the stressors (and by implication the resultant stresses) occur. The *ecological dread* would refer to the *necessity* that *all* the stressors occur together (in which case there is no doubt in the assessor's mind that E is likely to occur).
- Generally, it would not be known (at least at the outset) whether there is an additive, supra-additive ("synergistic") or infra-additive ("antagonistic") interaction among stressors. The way in which this is approached is largely a matter of assumption until further evidence is produced. The assumption will be reflected in the risk aggregation operators (t-norm and t-conorm in Eq. (3) below).

For the stressors noted above, these assumptions could be interpreted as:

- There exists a value of flow, q_0, in a given river section, for example, which will result in loss of sustainability if this flow is maintained for a sufficient period. Although the exact value is unknown, flow requirement studies (e.g. King and Louw, 1998) may yield some idea of what it might be. The flow-related concern and dread for any specific value of flow, q, under discussion, can be estimated from Eq. (2a):

$$\begin{aligned} Poss(E_Q) &= Poss((E \mid Q) \wedge Q) = t-norm\{Poss(E \mid Q), Poss(Q)\} \\ Nec(E_Q) &= t-norm\{Nec(E \mid Q), Nec(Q)\} \end{aligned} \tag{2a}$$

- There exists a critical value of toxic substance concentration, t_0, (as toxicity units) such that for any specific value t the toxics-related concern and dread would be given by Eq. (2b).

$$\begin{aligned} Poss(E_T) &= Poss((E \mid T) \wedge T) = t-norm\{Poss(E \mid T), poss(T)\} \\ Nec(E_T) &= t-norm\{Nec(E \mid T), Nec(T)\} \end{aligned} \tag{2b}$$

- Analogous to the above, the fuzzy critical habitat degradation value H is assessed by expert opinion so that for any specific level of habitat degradation, h, the habitat-related concern and dread will be given by Eq. (2c).

$$\begin{aligned} Poss(E_H) &= Poss((E \mid h_0) \wedge H) = t-norm\{Poss(E \mid H), Poss(H)\} \\ Nec(E_H) &= t-norm\{Nec(E \mid H), Nec(H)\} \end{aligned} \tag{2c}$$

The fuzzy set X is normalised since by assumption a stressor is only defined as such if there is at least one value of x such that $\mu_X(x) = 1$, i.e. there is at least one value for which E is entirely possible. Hence, the equivalence of the membership function values with the possibility of X.

A further result of the assumptions above is that the ecological concern ρ_c and ecological dread ρ_d is expressed in Eq. (3):

TABLE 1
Some possible *t-norms* and *–conorms* (Kruse et al., 1994) for use as aggregation operators on quantities *a* and *b* in assessing ρ_c and ρ_d

Type	t-norm	t-conorm	Implication ($\alpha \to \beta$)	Interpretation
Min-max(a,b)	$\text{Min}\{a,b\}$	$\text{Max}\{a,b\}$	$\text{Min}\{1-\alpha+\beta,\ 1\}$	Components contribute independently
Lukasiewicz(a,b)	$\text{Max}\{0,\ a+b-1\}$	$\text{Min}\{a+b,\ 1\}$	$\begin{cases} 1 \ \textit{if}\ \alpha \le \beta \\ \beta \ \textit{otherwise} \end{cases}$	Components additive
Probabilistic(a.b)	$a.b$	$a+b-ab$	$\begin{cases} \dfrac{\beta}{\alpha} \ \textit{if}\ \beta < \alpha \\ 1 \ \textit{otherwise} \end{cases}$	Intermediate between min-max and Lukasiewicz

$$
\begin{aligned}
\rho_c = Poss(E) &= Poss(E_Q \vee E_T \vee E_H) = t-conorm\{Poss(E_Q), Poss(E_T), Poss(E_H)\} \\
&= \min\left\{ \sum_{X \in \{Q,T,H\}} Poss(E_X), 1 \right\} \\
\rho_d = Nec(E) &= Nec(E_Q \wedge E_T \wedge E_H) = t-norm\{Nec(E_Q), Nec(E_T), Nec(E_H)\} \\
&= \max\left\{ 0, \sum_{X \in \{Q,T,H\}} Nec(E_X) \right\}
\end{aligned}
\tag{3}
$$

The implication is that if $\rho_c = 0$ then E is considered impossible (inasmuch as our knowledge base allows for that) and $\rho_d = 0$ by definition. If $\rho_c = 1$, then E is considered entirely possible (of course not necessarily entirely probable) and ρ_d may be ≥ 0, which means that not only is E possible, but it may also necessarily occur. If $0 < \rho_c < 1$, then E is possible to the extent ρ_c but $\rho_d = 0$ (if an event is not entirely possible it cannot be at all necessary).

The choice of *t-norm* and *t-conorm* in Eq. (3) for the stress aggregation needs to take cognisance of the knowledge about the interaction among stresses. For toxic substances, true synergism among the substances appears to be rare (Hermens et al., 1984a; 1984b; Calamari and Vighi, 1992) although it has been reported (Broderius and Kahl, 1985). Additivity of toxicity occurs more often than true synergism or supra-additivity. For other stressors, effects such as additivity have not been reported on if they do exist. Even less so has synergism among diverse stressors been reported on.

It is likely that additivity of effect among diverse stressors reflects the worst case, while additivity may also be possible. Some of the possible *t-norms* and *–conorms* that could be used in aggregating fuzzy risks are listed in Table 1.

For the aggregation of concern and dread (Eq. (3)) the Lukasiewicz aggregation with the implied additivity of stresses appears to the most conservative option. For the aggregation of risk components (Eqs. (2a) to (2c)), exposure and effect may be seen as contributing independently to the likelihood of effect, and consequently, the min-max aggregation would be more suitable.

End-point projection

The regulatory end-point E, which is at ecosystem scale, is unlikely to have data at the correct spatial and temporal scale from which it can be derived. It is more likely that, on a case-specific basis, phenomena at smaller spatial and temporal scales will be used to infer the occurrence of E. Lower level phenomena such as the disappearance of key species, loss of integrity, mortality of selected species are more likely to be used to infer E.

For example, assume that a toxic substance is introduced into a river system. From toxicity assessment it might be established that if the concentration of the toxic substance is x then the cumulative probability of an individual in a population of the test species Z will die, is y, with confidence interval (y_1, y_2). The toxicity concern, $Poss(E_T)$, and dread, $Nec(E_T)$, must be estimated from these data. In order to do this, it is necessary to follow some conceptual inference model such as Eq. (4)

Rule base (R):

IF concentration IS x THEN an individual of species Z IS dead (Possibility = y_1)
IF an individual of species Z IS dead THEN the population of Z IS lost (Possibility = α)
IF the population of Z IS lost THEN a key species IS lost (Possibility = β) (4)
IF a key species IS lost THEN integrity of the ecosystem IS irreversibly compromised (Possibility = γ)
IF integrity of the ecosystem IS irreversibly compromised THEN sustainability IS lost (Possibility = δ)
Observation (X): The concentration IS x (Possibility = ε)

An analogous rule base can be formulated for $N(E_T)$. The value of *Poss* (E_T) derives from the conjunction $R \wedge P$. This value will be a function of y_1, α, β, γ, δ and ε. In its simplest form *Poss* $(E_T) \leq \min\{y_1, \alpha, \beta, \gamma, \delta, \varepsilon\}$. (For $Nec(E_T)$ the inequality will be replaced by an equality.) This would support the notion that the possibility that toxics cause a loss of sustainability can be no stronger than the weakest inferential link. Since specific data for their assessment is usually lacking, the values for α, β, γ, δ and ε may conservatively be set equal to 1. The assumption should not simply be made that the confidence in the lower level phenomenon is equal to that of E (Suter, 1993; 1995).

Stressor-response relationships

A crucial component of the individual stressor concern (or dread) assessment is the conditional term *Poss* $(E|x_0)$ or $Nec(E|x_0)$. These terms are essentially the output of the effect assessment phase of an ecological risk assessment in the context where an end-point is fixed. It summarises the knowledge about the expectation of effect of the stressor on the system being assessed and answers the implied question: "What if the system is being exposed to the stressor"? In the present context, both E and x_0 are fuzzy entities and, hence, the condition term represents a fuzzy relationship, R_X. R_X is the formalised knowledge base on the relationship between the likelihood of E and x. The likelihood of individual stresses is derived from R_X and an observation X by Eq. (5). An expression for $Nec(E_X)$ can be similarly derived from Eq. (1).

$$
\begin{aligned}
Poss(E_X) = R_X \overset{t}{\circ} X &= \sup_x \{t - norm\{\mu_X(x), R_X(E,x)\}\} \\
&= \sup_x \{\min\{\mu_X(x), R_X(E,x)\}\}
\end{aligned}
\tag{5}
$$

The relationship R_X derives from a rule-base of the kind "If $X=x$ then $E=\varepsilon$" where the truth-value of $X=x$ is $\mu_X(x)$ and that of $E=\varepsilon$ is $\mu_E(\varepsilon)$. This can then be formulated as "$\mu_X(x) \rightarrow \mu'_E(\varepsilon)$". Using the max-min implication (Table 1) Eq. (5) becomes Eq. (6).

$$Poss(E_X) = \sup_x \left\{\min\left\{\mu_X(x), \min_\varepsilon\{1-\mu_X(x)+\mu_E(\varepsilon), 1\}\right\}\right\} \quad (6)$$

Evaluating R_X now becomes the problem of evaluating the relationship $\mu_X(x) \rightarrow \mu_E(\varepsilon)$, or "IF $\mu_X(x)$ THEN $\mu_E(\varepsilon)$". There are two distinct ways to generate this assessment:

- Cause to effect: Given a stressor value x, to what extent will its impact comply to the description E (i.e. $x \rightarrow E$) and
- Effect to cause: Given a level of effect ε, what are the levels of x that correspond to ε (i.e. $E \rightarrow x$).

In general, this need not be a mathematical-functional relationship. If the best knowledge available is in the form of fuzzy "rules" such as those in Eq. (4), then the stressor-response relationship (SRR) is at best a fuzzy mapping of the stressor value domain to the response likelihood domain.

Hypothetical case study

In an ecological risk assessment study, it is agreed that there are three major stressors in a catchment, i.e., unidentified toxic substances, deviation from expected flow and physical habitat degradation. There are three types of information that is required from expert input:

- Definition of the SRR from a) the lowest stressor value where effect E may be expected to be discernable (x_{11}), b) the lowest stressor value where E may be entirely possible (x_{12}), c) the highest level where E may be discernable (x_{21}) and d) the highest level where E is entirely possible (x_{22}).
- The epistemological confidence on projecting from the observable response to the regulatory end-point (α, β, γ, δ and ε).
- The likelihood of the occurrence of the stressor. ($\mu_X(x)$)

Fuzzification of concern and dread

Consider a situation in a river system where the critical effect, E, being assessed is "loss of sustainability". Due to the epistemic uncertainty relating to mechanisms, thresholds, subjectivity in assessments, etc. in a river system, the risk of E (expressed here as the possibility of E) is described in terms of categories rather than numerical terms. For example, the level of risk may be assessed as belonging to a class K such that the set K = {Insignificant, Low, Marginal, Significant and High} as shown in Fig. 1.

These classes are vague since their boundaries may be a matter of interpretation. An effect possibility of 0.2 might be described as being 'low' or 'marginal' to some extent. Consequently, the classes are modelled as fuzzy sets. These same 'fuzzification' parameters might also be used in describing the concern and dread levels since they deal with the same type of possibilistic measures.

The definition of individual stressor effect possibility (Eq. (6)), as well as the aggregated concern values (Eq. (3)), ensure that at least one of the fuzzy sets will have a membership value of 1. This means that it will be possible to describe the concern level in a river or stream in terms of at least one of the classes. However, it may be possible that more than one class has a membership of 1, in which case the worst class that has a membership of 1 will logically be class descriptor for the river situation.

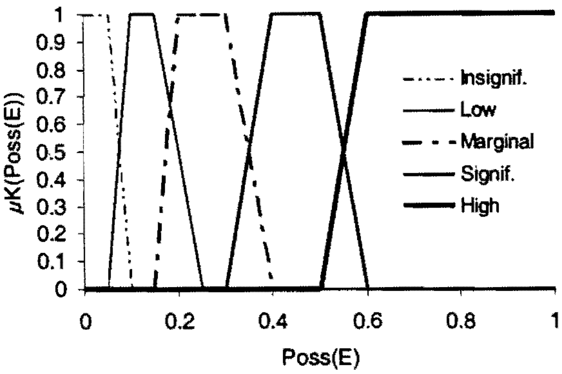


Figure 1
The parameters for describing the possibility of E *in terms of the set* K *of fuzzy labels. The fuzzy set is defined by the degree to which the possibility of effect,* Poss*(E), corresponds to the descriptor* K.

Toxic stress

The toxicity stress is determined by toxicity bioassessment studies without specifically identifying the toxic components and is expressed as toxicity units. A toxicity unit is defined as 100 divided by a benchmark effect concentration expressed as a percentage of the effluent. The data are derived from single species toxicity tests and projections of effect to population level (e.g. Caswell, 1989). The no-observable-effect concentration (NOEC) is taken to be at 10% of the EC50.

In-stream objectives of 0.3 TUa and 1 TUa have been suggested as levels where no critical effects should be observed (USEPA, 1991, Tonkes and Balthus, 1997) and these values are used for x_{11} and x_{21} respectively. It is assumed that at double the EC50, sustainability might be lost if predation pressure is high while, even under the best circumstances, sustainability is in jeopardy if 99% (corresponding perhaps to 3 times the EC50) of a population dies. These values are used for x_{12} and x_{22} respectively (Fig. 2(a)). The possibility distribution for x is assumed to be a triangular distribution such that $\mu_X(x) = 0$ corresponds to the 5[th] and 95[th] percentile values while $\mu_X(x) = 1$ corresponds to the median value. The values of α, β, γ, δ and ε in Eq. (4) are all assumed to be 1.

Flow stress

The flow stress, q, is assumed to be due to the reduction of the expected flow in stream. The value of $q = 0$ when the stream flow is very similar to pristine flow while $q = 1$ corresponds to critical disruption of stream flow. The values for the mapping parameters are entirely hypothetical (Fig. 2(b)).

Habitat stress

The habitat degradation is assessed by a river ecologist and expressed as a percentage deviation from what is expected to be pristine. The values for the mapping parameters are entirely hypothetical (Fig. 2(c)). The fuzzy relationships were assumed to show a triangular distribution such that for any stressor level, the effect is given by a triangular distribution with its least likely values given by y_1 and y_2 (see Appendix) and its most likely value by y_m.

TABLE 2
The scenarios in which the ecological risk assessment is evaluated.

Scenario	Toxic substance status	Flow status	Habitat status
1	The levels are practically pristine. Discharges are mostly assimilated	Very little abstraction or water loss is evident. Sporadic abstraction has a minor impact.	Practically pristine. Only minor modifications (10%) are found.
2	Substantial discharges exist. With a very low frequency up to 5 TUa is found while there is usually some chronic toxicity detected (0.1 TUa). Values of 1 TUa is found commonly.	Extensive abstraction takes place at times resulting stressor levels within 20% of critical levels. On rare occasions the flow is practically pristine, but mostly the flow is within 50% of pristine.	There is almost no pristine habitat left with some areas being largely modified (about 75%) while most of the stream has about 50% suitable habitat left.
3	Rigid control on point sources is instituted but on rare occasions 1 TUa is still found, but mostly toxicity is around 0.3TUa or even as low as no detectable toxicity.	Some control on abstraction is possible and flows within 20% of expected can often be achieved. However, on rare occasions up to 80% of the pristine flow is abstracted.	Some habitat remediation could be effected so that most of the river now has 25% loss of the pristine habitat while the worst case has only about 50%.
4	Toxicity is managed to be around 0.55TUa most of the time while excursions up to 1.1 are rarely found.	Same as in 1.	Same as in 1.

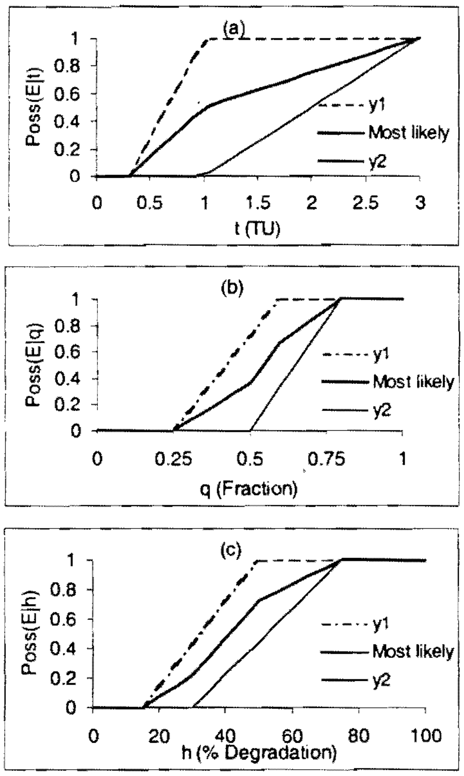


Figure 2
The fuzzy mapping representing the SRR's for the stressors in this study: (a) SRR for toxicity stress, (b) SRR for flow stress and (c) SRR for habitat stress.

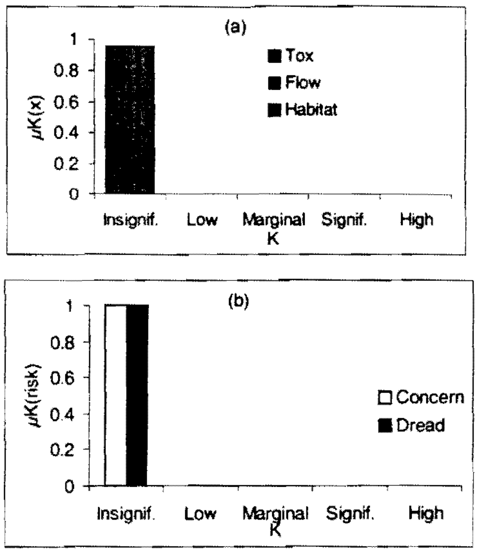


Figure 3
The classification of (a) the stressor specific possibility of effect in terms of fuzzy set membership to the class K (see Fig. 1) and (b) the concern and dread for Scenario 1 (Table 2).

Methodology

Eqs. (1) to (3), (5) and (6) as well as those in the Appendix were solved using an Excel97 spreadsheet under Windows 95.

The use of ecological concern and ecological dread was investigated by considering its value in four scenarios as described in Table 2.

The narrative description of scenario 1 in Table 2 yields

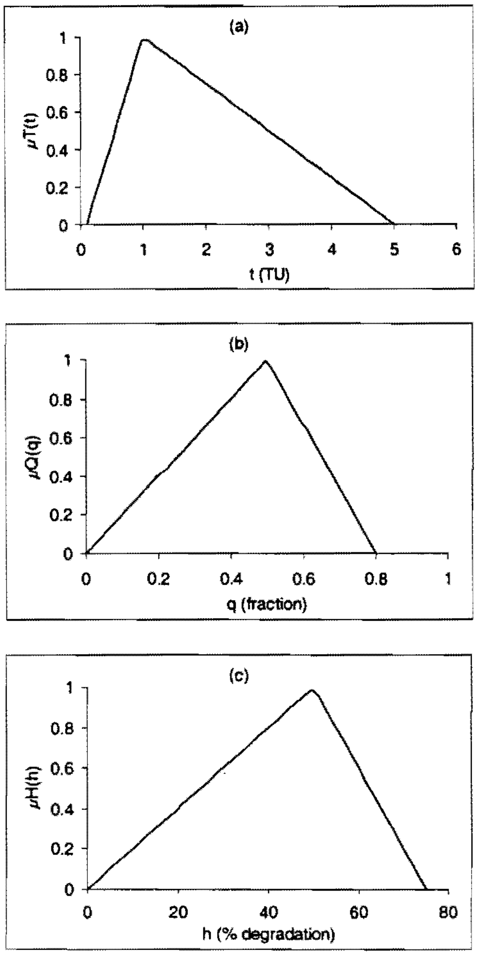


Figure 4
The stressor possibility distributions for (a) toxics-, (b) flow- and (c) habitat-related stress (expressed as $\mu_x(x)$) derived from the descriptive data for Scenario 2 in Table 2

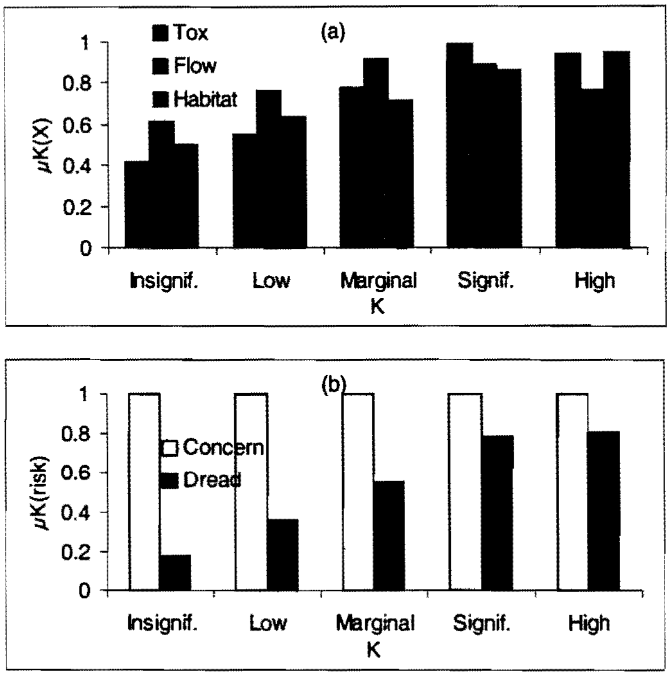


Figure 5
The classification in terms of class K (Fig. 1) of (a) stressor specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 2

stressor possibility distributions that are triangular with vertex at (0,0). The stressor possibility distributions for scenario's 2 and 3 are shown in Figs. 4 and 6 while the SRR's are shown in Fig. 2.

Results and discussion

The individual stressor risks are shown in Figs. 3, 5, 7 and 8.

Scenarios 1 to 3 were chosen to represent a pristine, a heavily utilised and a reasonably managed system respectively. The pristine system, not surprisingly, yielded an assessment of insignificant risk for each individual stressor (Fig. 3(a)). Consequently, both the concern and dread (Fig. 3(b)) are 'insignificant' as would be expected.

In the case of the heavily utilised system (Fig. 5) the individual stressor risk values are either 'significant' (toxics and flow) or 'high' (habitat), considering the maximum membership values. The aggregation method used here results in a concern membership value of 1 to all classes. Since the worst class will reasonably dominate, it could be said that the concern level is 'high'. In this case the dread value is used to distinguish between the classes, so that a dread class of 'high' could be allocated.

An analysis of the stressor risk contributions in Scenario 2 shows that all the stressors need attenuation. It is assumed that in the managed system (Scenario 3) it is possible to manage the discharge of toxics as well as abstractions to a reasonable extent while stream habitat remediation is less successful (Fig. 6). The results (Fig. 7) indicate that although toxic and flow risk are now largely 'insignificant' and habitat risk is 'low', on aggregate the concern level is still no better than 'high'. The dread value though has become ''insignificant', demonstrating that progress had been made in improving the situation.

Scenario 4 (Fig. 8) was used to illustrate a possible use of concern and dread assessment in assessing the change in criteria (in this case the example of toxicity management criteria). It was now assumed that both habitat and flow risk were insignificant. By systematically changing of the most likely value and the upper limit value in the toxicity possibility distribution, it was attempted to find a parameter set that would be on the verge of changing the concern assessment from 'insignificant' to 'low'. This parameter set is reflected in Table 2. This is in spite of the toxic effect possibility being 'low' or even 'marginal'.

The interpretation of Scenario 4 is that if there are no other stressors that could significantly contribute to the ecological risk, then the parameter values for this scenario will be the maximum allowable to maintain 'insignificant' concern and dread levels.

It has been assumed that risk objectives for the river have been set. This is generally not true for South African rivers. The parameters (i.e. the *Poss*(E) values defining the fuzzy set trapezium in Fig.1) used for classifying response possibility are critical. In this hypothetical study the fuzzification as depicted in Fig. 1was simply assumed. No formal procedure was put forward to derive rational values for these parameters and this aspect needs more

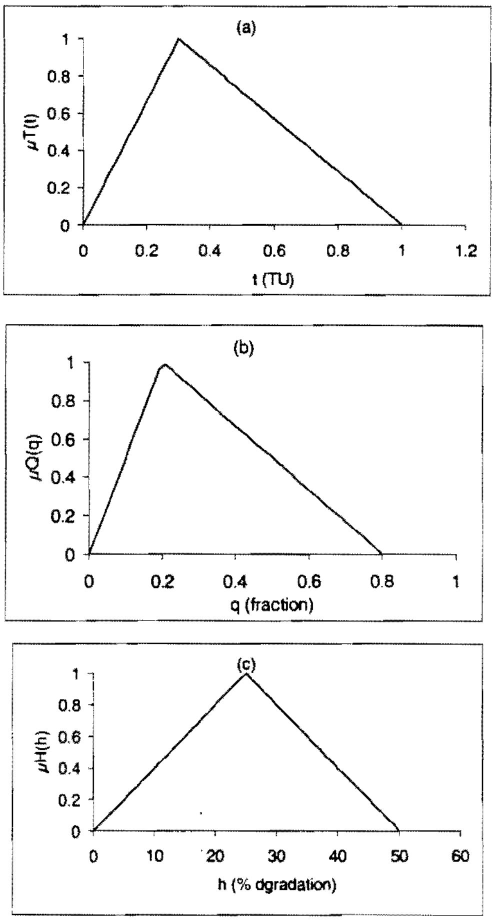


Figure 6
The stressor possibility distributions for (a) toxics-, (b) flow- and (c) habitat-related stress (expressed as $\mu_x(x)$) derived from the descriptive data for Scenario 3 in Table 2

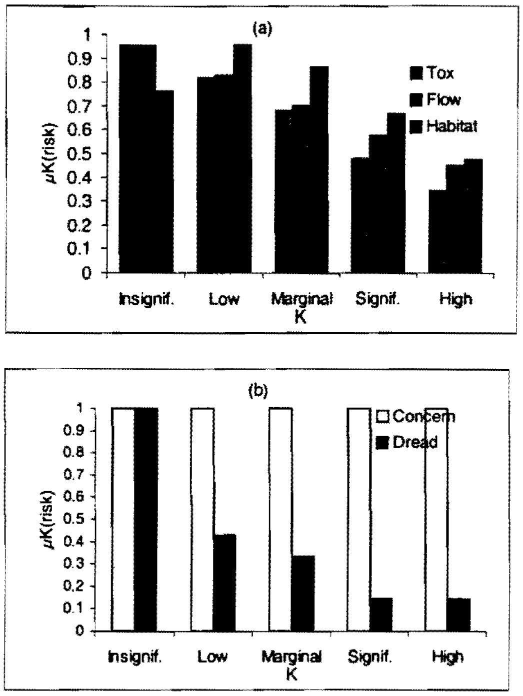


Figure 7
The classification in terms of class K (Fig. 1) of (a) stressor-specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 3

extensive consideration. Any procedure for deriving the fuzzification parameters would have to take cognisance of:

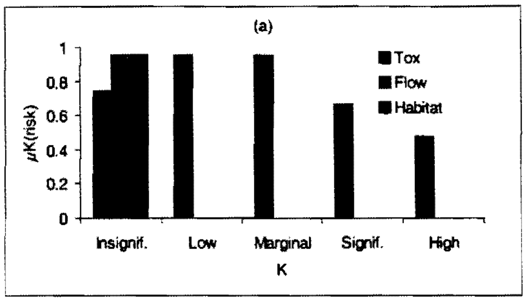
- correspondence between observed system assessments and the concern and risk classes projections, and
- the risk perceptions of the user community.

The former problem can probably best be addressed by analysis of a database containing both bio-monitoring and stressor data by a tool such as neural networks. The assumption is that the concern and dread levels will generally be reflected in the trends in stream bio-integrity. The latter problem is similar to the domain of risk communication except that risk values are usually in probabilistic rather than possibilistic terms.

The concern and dread assessments are also significantly affected by the SRRs. The use of fuzzy mapping as SRRs addresses this problem to some extent. With reference to SRRs it is noted that:

- If the uncertainty in the different SRRs differ widely, it is apparent that the higher uncertainty will dominate the assessment uncertainty. It may, for example, be unnecessary to insist on high confidence toxicity response data (simply because it can be achieved), while having to accept very coarse data on habitat related stressor-response information.
- It has tacitly been assumed that the identification of a stressor had taken into consideration a temporal component if at all applicable. It is known that toxic substances may accumulate over a period to toxic levels in an organism (e.g. Mancini, 1983; Legierse, et al., 1999). However, for toxic substances intra-organismal stressor exposure was assumed to be proportional to the stressor magnitude, while the temporal characteristics of the stressor had been neglected.
- In the case of flow stress, the assumption that stress is simply proportional to reduction from expected flow, is probably too simplistic. It is known that a certain amount of flow variability is both normal and necessary for the functioning of most South African aquatic ecosystems (King and Louw, 1998). A more realistic description of flow-related stress would likely involve a stochastic variable whereby the range becomes abnormal.
- The duration of stress has not been explicitly addressed for any of the stressors. This paper does not particularly concern itself with the detail of such a description, except to postulate that such a descriptor will have a magnitude component and a temporal duration component, both of which could be variable. It is possible that the variables used to characterise the stress descriptor would be crisp, but the advantage of the fuzzy approach is that they could be vague or fuzzy (depending on the state of knowledge) without invalidating the approach.

Considering Eqs.(2), (3) and (5) or (6), it is trivial to recognise that there are theoretically an infinite number of stressor-specific fuzzy risk combinations that result in the same concern (or dread) value. If only a single stressor was being addressed, it would simply



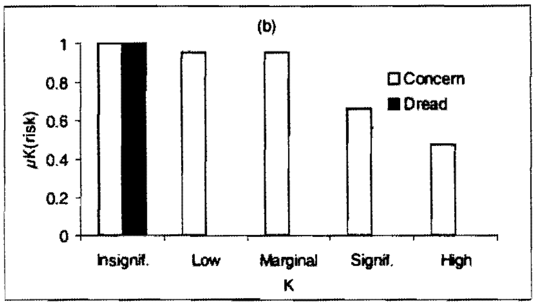


Figure 8
The classification in terms of class K (Fig. 1) of (a) stressor-specific effect possibility (Poss(E)) and (b) concern and dread for Scenario 4

require a waste load allocation-type of calculation to distribute the stressor load among stressor sources (USEPA, 1991). However, the essence of the concern and dread calculation is the aggregtion of the diverse stressors into one measure. This means that in order to select among the infinite number of source-specific stressor-level combinations, some form of optimisation procedure would be called for. While this is a more complex task than a waste load allocation (USEPA, 1991) it also increases the management flexibility by opening the way for cost-risk-benefit calculation. This aspect requires some investigative work, although there is a substantial volume of work in the field of fuzzy optimisation (Dubois and Prade, 1994, Klir and Yuan, 1995, Sasikumar and Mujumdar, 1997).

The mathematical structure of the model is unaffected by the number of premises and propositions since it is based mostly on *max* and *min* operations. The extension to additional interactions is trivial. However, the possibility and necessity measures for the rules need to be stated as they determine the confidence in the overall assessment and this holds true for the stressor-effect implications.

Conclusions

This paper is an attempt to motivate the use of a possibilistic approach to ecological risk assessment rather than the more common probablistic approach in cases where there is epistemic uncertainty as well as stochasticity in the system being assessed. The use of fuzzy logic and a possibilistic approach to ecological risk makes use of three types of information:

- an assessment of the relationship between stressors magnitude and the expected response at a suitable level of organisation in the form of a fuzzy implication relationship,
- a possibility distribution for each stressor, and
- a logical inference model connecting direct stressor effects and the higher level end-points for the assessment in the form of a rule base.

The possibilitic ERA formulation has the advantage that it could make use of the vague information that is often all that is available for ecosystems effects, but it can also be used where precise information is available. For an application where there is no need for more precise or numeric risk data, this fuzzy set approach may be sufficient. However, the use of fuzzy variables cannot be used as a cover for bad or misleading data. The scientific quality of data is a separate issue from fuzziness. While high quality data can be fuzzified, doubtful, vague or conflicting data cannot be improved by this technique. It is necessary to be explicit with the uncertainty and vagueness in the formulation of the ecological risk assessment problem.

The parameters used in the fuzzification of data need to be considered carefully. These must be agreeable to both the risk assessor and the risk manager. This is particularly crucial where stressor response curves are very steep, i.e. where large changes in response (or fuzzy set) correspond to relatively small changes in stressor exposure.

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Appendix

The stressor response relationship for each stressor is delineated by a fuzzy mapping (See Figs. 2 to 4) such that:

$$y_1 = \begin{cases} 0 & if\ x \le x_{11} \\ \dfrac{x - x_{11}}{x_{12} - x_{11}} & if\ x_{11} < x < x_{12} \\ 1 & if\ x \ge x_{12} \end{cases}, \quad y_2 = \begin{cases} 0 & if\ x \le x_{21} \\ \dfrac{x - x_{21}}{x_{22} - x_{21}} & if\ x_{21} < x < x_{22} \\ 1 & if\ x \ge x_{22} \end{cases} \quad \text{and} \quad y_m = \frac{y_1 + y_2}{2}$$

$$\text{so that } \forall y \in [y_2, y_1] Poss(E) = \begin{cases} 0 & if\ y \notin [y_1, y_2] \\ \dfrac{y - y_2}{y_m - y_2} & if\ y_2 \le y \le y_m \\ \dfrac{y_1 - y}{y_1 - y_m} & if\ y_m < y \le y_m \end{cases}$$

where y is the possibility distribution of the effect E derived from the mapping. The membership of y to class L, μL(y), where class L is described by a trapezoidal function such that:

$$\mu_L(y) = \begin{cases} 0 & if\ y < a\ or\ y > d \\ \dfrac{y - a}{b - a} & if\ a \le y \le b \\ \dfrac{d - y}{d - c} & if\ c \le y \le d \\ 1 & if\ b < y < c \end{cases}$$

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ECOLOGICAL CONCERN AS A FACTOR IN THE OPTIMAL ATTENUATION OF DIVERSE STRESSOR SOURCES IN A STREAM.

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ABSTRACT

The use of an objective function based on fuzzy ecological effect expectation in a genetic optimisation algorithm to obtain site or situation specific stressor attenuation values for the management of diverse stressors emanating from several sources, is investigated. The approach is based on the premise that both regulator and regulatee are able to formulate their goals in fuzzy terms. In the case of the regulator the goals will be formulated in terms of acceptability of levels of ecological concern (a fuzzy analogy to ecological risk). In the case of the regulatee it will be formulated in terms of acceptability of the level of attenuation, which is also the control variable. A hypothetical catchment is used to illustrate the principle.

KEYWORDS

Fuzzy risk; genetic algorithm optimisation; impact assessment; impact management

INTRODUCTION

Sustainability of the aquatic environment is a water resource management goal required by many countries including South Africa. Common water resource management problems in the attainment of this goal, exacerbated in a relatively poor, water-scarce country, include:

1) integrating the impact of diverse stressors which result in the same high level effect such as loss of sustainability,
2) setting goal-related management objectives for such stressors, recognising technological or technology related constraints, and
3) the need for an equitable and transparent apportionment of impact reduction among the users of the resource.

A potential conflict between the regulatory agency charged with the protection of the resource and the users intent on using the river capacity to the full, results in pitting an apparently ethereal concept against material realities. The second and third problems are typically addressed by waste load allocation (when stressor specific numeric criteria are available) and multiple objective optimisation, both of which may entail a stochastic approach (Lohani and Thanh, 1978; Burn and McBean, 1985; Chadderton and Kropp, 1985; Burn and Lence, 1992; Hutcheson, 1992; Tung, 1992; Cardwell and Ellis, 1993; Lung, 1995) or a fuzzy approach (Hathhorn and Tung, 1989; Sasikumar and Mujumdar, 1997). The two major components that add to the stochasticity have been considered to be the variability in river and effluent flow. However, resource management to ecological goals is further complicated by:

- variability of susceptibility to stressors within and among various levels of ecological organisation,
- uncertainty introduced by insufficient system specific knowledge and,

- vagueness relating to various ecosystem-level characteristics such as integrity and sustainability (Karr and Dudley, 1981;Cairns and Niederlehner, 1995; Karr, 1996; Ludwig, et al., 1997; USEPA, 1997).

BACKGROUND

It could be argued that the main problems in solving the problem of the apportionment of impact abatement relates to a) the expression of the aggregated impact of diverse stressors and b) a formulation of the optimisation problem that is based on a common objective for resource protector and user.

The diverse-stressor problem

An expression of risk, ρ, is proposed which is epistemic of the likelihood that the system will succumb to the end-point E: loss of sustainability. Loss of sustainability is here viewed as a fuzzy end-point, which in most real cases may only characterised by qualitative, possibly vague, descriptors. As such, ρ expresses an assessment, based on available evidence, that this end-point will be attained through any stressor. This likelihood is dependent on the likelihood of the occurrence of the stressors and the likelihood of the E conditioned on the magnitude of stressors. For example, assume stressor values corresponding to E for flow-related stress, toxic substance-related stress and habitat-related stress are grouped in sets Q, T and H respectively. If the likelihood is expressed in terms of possibility, then ρ could be expressed as Eq. [1], where $\Pi(.)$ denotes a possibility measure. This possibilistic analogue of ecological risk is here referred to as ecological concern.

$$\rho = \Pi(T \cup H \cup Q) = t\text{-}conorm\{\mu_T(t),\ \mu_H(h),\ \mu_Q(q)\}. \qquad [1]$$

The right hand side of Eq. [1] is derived by considering a toxicity value t, a habitat degradation value h and a flow stress value q (with set membership functions $\mu_T(t)$, $\mu_H(h)$ and $\mu_Q(q)$ respectively), occurring in the system. The possibility that sustainability will be lost due to this set of circumstances will be expressed by ρ. In this study the *max* operator had been used to express the *t-conorm*, but a number of other operators (including the probabilistic sum) are available to tailor the operation to the situation being modelled (Klir and Yuan, 1995). The membership stressor value x to fuzzy set X has been estimated from:

$$\mu_\Phi(\phi) = \min_{\phi}\{\pi_{E|\varphi}(\phi), \pi(\phi)\} \qquad [2]$$

where $\Phi \in \{T, H, Q\}$, $\phi \in \{t, h, q\}$ and $\pi_{E|\phi}(\phi)$ and $\pi(\phi)$ are the possibility distribution of loss of sustainability conditioned on the stressor value ϕ and the possibility distribution of the stressor ϕ respectively.

Combining Eqs. [1] and [2] yields the well-known max-min composition of possibility theory (DuBois and Prade, 1994; Klir and Yuan, 1995).

$$r = \max\left\{\min_{\phi}\{\pi_{E|\phi}(\phi), \pi(\phi)\}\right\} \qquad [3]$$

Ecological concern as used here expresses the maximal expected possibility of a vague end-point (i.e. the loss of sustainability in this case). At the ecosystem level, where specific information is often sparse, expert opinion may be needed to establish, not only at what point sustainability is considered to be lost, but also to formulate the stressor response relationship. It may not be possible to stipulate any more than an expected no-effect or threshold of effect level and an expected unacceptable effect level.

Formulating the optimisation problem

The common ground between regulator and user may be found in the level of satisfaction, λ with the regulated situation. The objective for optimisation may be expressed as:

$$\text{Max } \lambda$$
$$\text{s.t.} \quad \lambda = \begin{cases} 0 & \text{if } \lambda_R < \zeta \\ \min\{\lambda_R, \lambda_x, \lambda_{eq}\} & \text{if } \lambda_R \geq \zeta \end{cases} \qquad [4]$$
$$x_{ij} \geq 0$$

and λ_R, λ_x and λ_{eq} are defined below.

Consider the situation where stressor i ($i \in \{1,.., n\}$) is introduced at j ($j \in \{1, .., m\}$) different points. On the part of the regulator λ will be determined by satisfaction of the management objective: $\rho \leq \rho'$ where ρ' is the concern (or risk) objective for the water body. While it would be ideal to have a crisp value for ρ', it might also be a fuzzy number not necessarily symmetric around ρ'. The concern objective may be described by ρ^{min} and ρ^{max}, levels below which the concern is perfectly acceptable and above which it is completely unacceptable respectively. The overall satisfaction with respect to the concern objective is indicated by λ_R. The values of ρ^{min} and ρ^{max} may be, in general, reach specific. Downstream of each point j, there may in principle be a different degree, $\lambda_{r,j}$, to which the concern objective $(\rho_j^{\max}, \rho_j^{\min})$ is satisfied. The level of concern, r_j, is the source specific concern calculated from Eq. [3] and $\lambda_{r,j}$ would a fuzzy set of Type 2 (Figure 1) on r_j and $\rho_j^{\min}$ and $\rho_j^{\max}$ as the minimum and maximum criteria respectively. As a matter of policy, it might be decided by the regulatory authority that a minimum concern satisfaction level ζ may be imposed (i.e. if the ecological concern exceeds ζ, λ =0 irrespective of other considerations). For this study ζ = 0 was assumed.

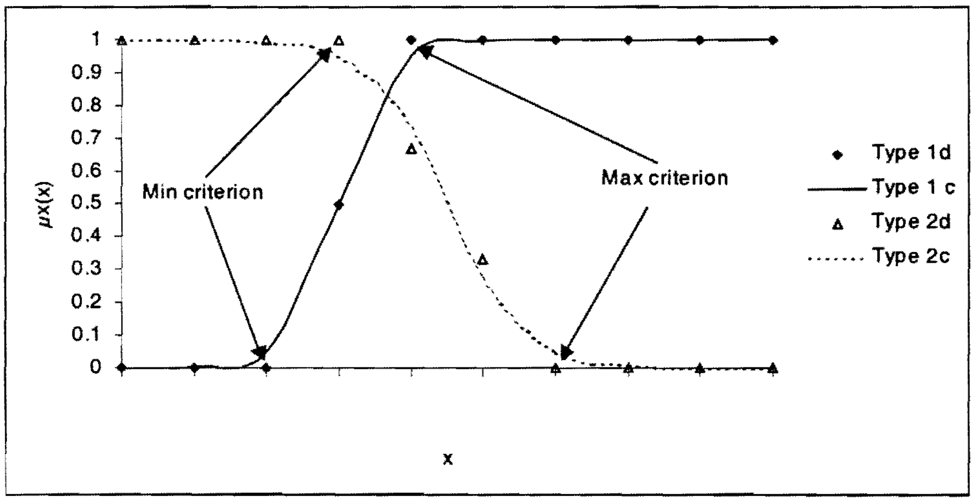

Figure 1. An illustration of the two types of fuzzy set membership functions used in this study. The d(iscrete) and c(ontinuous) versions are shown.

On the part of the regulatee, λ will be determined by stressor source management issues, specifically the acceptability of *stressor reduction* criteria for stressor sources. It is assumed that the control variable is the stressor attenuation level x_{ij} for stressor i from source j ($x_{ij} \in [0, 1]$, where no reduction implies $x_{ij} = 0$ and complete stressor removal implies $x_{ij} = 1$) for stressor i from source j. A stressor reduction level $x_{ij} > 0$ imposes a burden on the source management agency (which may be in the form of the treatment cost or some other direct or indirect operational problem). The satisfaction of each stressor source combination is indicated by λ_{ij}.

It is assumed that a crisp attenuation acceptability criterion would not be feasible and that the fuzzy equivalent can be formulated as a fuzzy set from an acceptability pair $\{x_{ij}^{min}, x_{ij}^{max}\}$ from each stressor source manager. These acceptability criteria may incorporate source- and stressor-specific weighting of cost and technological implications of a treatment level x_{ij}. Here, x_{ij}^{min} represents a treatment level that is completely acceptable, while x_{ij}^{max} represents a treatment level which, for whatever reason, is completely unacceptable. The value of λ_{ij} is a fuzzy set of type 2 (Figure 1) on xij with , x_{ij}^{min} and x_{ij}^{max} the minimum and maximum criteria respectively.

Conservatively, the value for the overall satisfaction with the regulated attenuation can be expressed as: $\lambda_x = \min\{\lambda_{ij}\}$.

The satisfaction of the requirement for equity in stressor attenuation among identical stressors is expressed as λ_{eq} is expressed as a fuzzy set of Type 2 (Figure 1) on the maximum difference (δ) in required attenuation among all stressors and sources (Eq. [5]).

$$\delta = \max_{i}^{m}\left(\frac{\max_{i}^{n}\{x_{ij}\} - \min_{i}^{n}\{x_{ij}\}}{(\max_{i}^{n}\{x_{ij}\} + \min_{i}^{n}\{x_{ij}\})/2}\right) \qquad [5]$$

Equity acceptability criterion values ε^{min} and ε^{max} of 0 and 0.2 respectively were used in this study.

METHOD

The application of this methodology is illustrated by a typical data set from a small stream in South Africa receiving water from small sewage treatment works while serving as irrigation water for smaller farms. Many such streams are at the headwaters of, or serve as refugia for major rivers. The stream is modelled as a set of four effluents and one abstraction (Figure 2). The river habitat characteristics downstream is associated with the node just upstream as part of the characteristics determining its ecological concern.

Both the stressor distribution and the site-specific conditional response ($\pi_{E|\phi}(\phi)$) are determined by expert input. The most difficult would seem to be the estimation of effect conditioned on stressor value. This is conceptually equivalent to a stressor-response relationship where the response is the epistemic possibility of observing the target effect. In all cases a minimum and maximum effect criterion (ϕ^{min} and ϕ^{max}) were elicited such that $\pi_{E|\phi}(\phi) = 0\ \forall\ \phi \leq \phi^{min}$ and $\pi_{E|\phi}(\phi) = 1\ \forall\ \phi \geq \phi^{max}$. All possibility distributions were then converted to continuous function of Type 1 (Figure 1) by Eq. [6a]

The toxic substance concentrations is expressed in terms of toxicity units which in this case had been derived from an extended chronic whole effluent laboratory toxicity assessment and population growth projection (e.g. Jooste and Thirion, 1999). Based on what is known about the biota in a stream section between nodes, as well as the relative sensitivity of the laboratory test organism compared to those biota, an assessment is made of the maximum and minimum toxicity effect criteria. The toxic substance has been assumed to be subject to pseudo first-order degradation kinetics (constant 0.2 day^{-1}) and dilution. The concentrations were calculated by simple mass balance based on interval arithmetic using α-cuts from the toxic substance- and flow possibility distributions ($\alpha = 0.05$). The parameters are shown in Table 1.

The flow-response relationship is estimated from querying experts to supply q^{min} and q^{max} while using some form of instream flow requirement methodology (e.g. King and Louw, 1998). The habitat related stress-response relationship is derived similarly. The stressor exposure possibility distributions were derived directly from the corresponding probability distribution by requiring that $\max(\pi(\phi)) = 1$.

Three scenarios are presented. The parameter values for scenario A are presented in Table 1. Reach independent concern acceptability criteria of 0.05 and 0.15 ρ^{min} and ρ^{max} were used. For scenario's B and C the toxics attenuation acceptability criteria for source 1 and concern acceptability criteria respectively were changed as shown in Table 2.

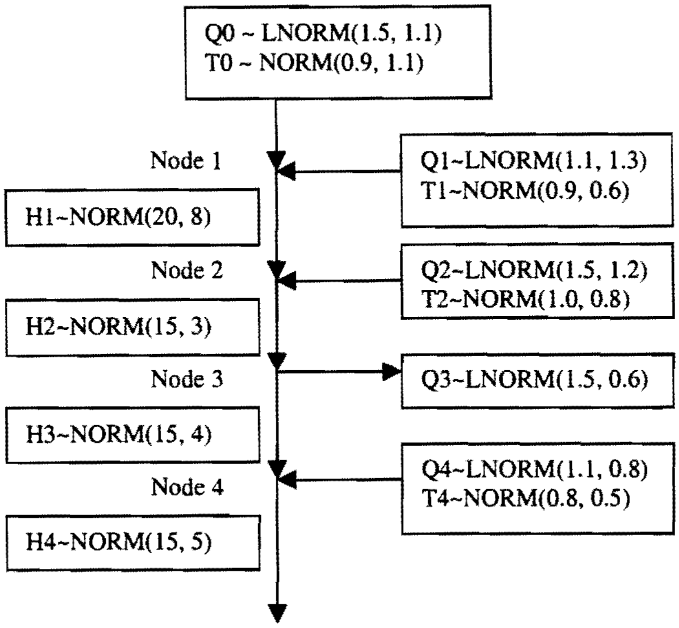

Figure 2. Schematic of the test input data used to illustrate the application of ecological concern-based optimisation. NORM (*a*, *b*) and LNORM(*a*, *b*) indicates normal and lognormal distributions respectively with median *a* and standard deviation *b*. The units for flow distributions (*Qj*) are megalitres per day, toxics distributions (*Tj*) are toxicity units and habitat degradation (*Hj*) are percent.

Table 1 Numerical input values for scenario A. (1 ML.day^{-1} = 0.0116m^3.s^{-1}.)

Parameter	Units	Node 1	Node 2	Node 3	Node 4
t^{min}	TUc	1.5	1.5	2	2
t^{max}	TUc	2.1	2.1	2.5	2.5
Flow stress effect min	ML.day-1	2.2	2.5	2.5	2.6
Flow stress effect max	ML.day-1	3.5	3.5	3.5	4
Habitat stress effect min	%	30	30	30	30
Habitat stress effect max	%	75	75	65	75
Retention time source to source	Days	2	3	2.5	4
x_q^{min}	-	-	-	0	-
x_q^{max}	-	-	-	0.6	-
x_t^{min}	-	0.2	0.2	-	0.3
x_t^{max}	-	0.7	0.8	-	0.75
x_h^{min}	-	0	0	0	0
x_h^{max}	-	0.15	0.1	0.1	0.2

Table 2. The changes in parameters associated with scenarios B and C

Scenario	$x_{t1}^{min}, x_{t1}^{max}$	ρ^{min}, ρ^{max}
B	0.01, 0.3	0.05, 0.15 (same as A)
C	0.2, 0.7 (same as A)	0.01, 0.05

The optimisation was performed using a genetic algorithm (Bäck, 1996) with search heuristics and focussing of search domain described in Ndiritu and Daniell (1999). A population of 20 solutions was used including the best four individuals from the previous generation, random crossover and a mutation rate of 0.01. The parents were selected randomly from an exponential probability distribution. After an epoch of 40 generations, 18 of the population were regenerated from an exponential distribution centred on the focussed search domain. A cycle of 10 epochs was repeated 10 times. In order to circumvent the problem of degeneracy of solutions in the optimisation heuristic, both effect and stressor distributions were modelled as the continuous approximations of the discrete sets (Figure 1). Type 1 and Type 2 continuous sets were expressed by either of Eqs. [6a] or [6b].

$$f(x) = \frac{1}{1 + a \cdot e^{-kx}} \qquad [6a]$$

$$f(x) = \frac{a \cdot e^{-kx}}{1 + a \cdot e^{-kx}} \qquad [6b]$$

Where the parameters a and k were calculated by considering the minimum criterion as corresponding to 0.05 (or 0.95 for type 2) and the maximum criterion corresponding to 0.95 (or 0.05 for Type 2).

RESULTS AND DISCUSSION

The acceptability levels are quite low (Fig. 3) when attenuation equity is required among stressor sources. The tightening of concern bounds (Scenario C, Fig. 3) results in higher attenuation levels for toxics and much lower satisfaction levels. Lowering the acceptability bounds for toxic attenuation for source 1 (Scenario B) has very little effect except to lower λ since the equity constraint tends to treat all toxics sources the same.

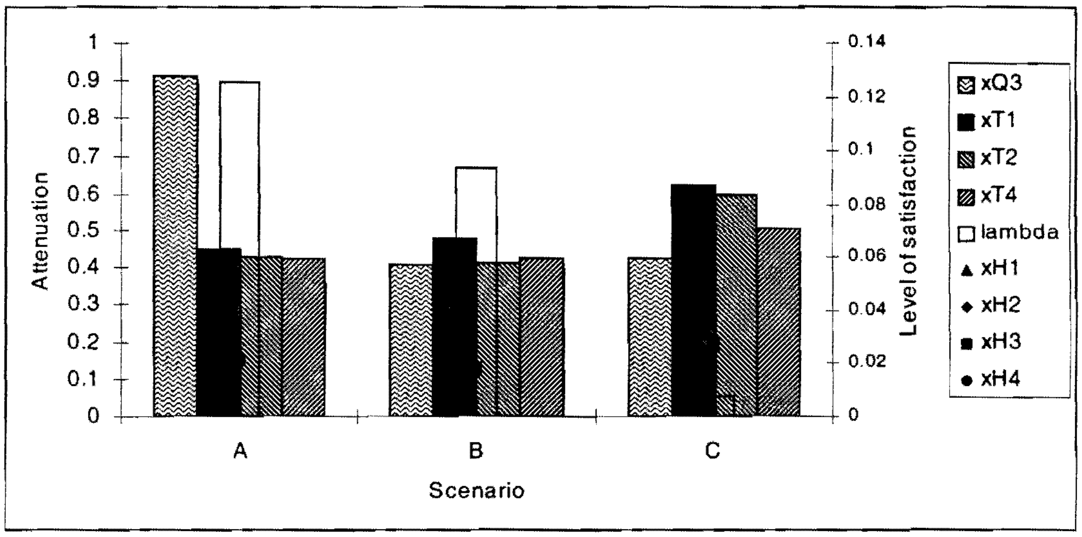

Figure 3. The attenuation levels (x) for flow (Q), toxics (T) and habitat (H) related stressors (for each of the 4 sources in the example) corresponding to the highest value of the overall acceptability λ. The value of λ is represented by the open rectangle and refers to the right hand ordinate axis.

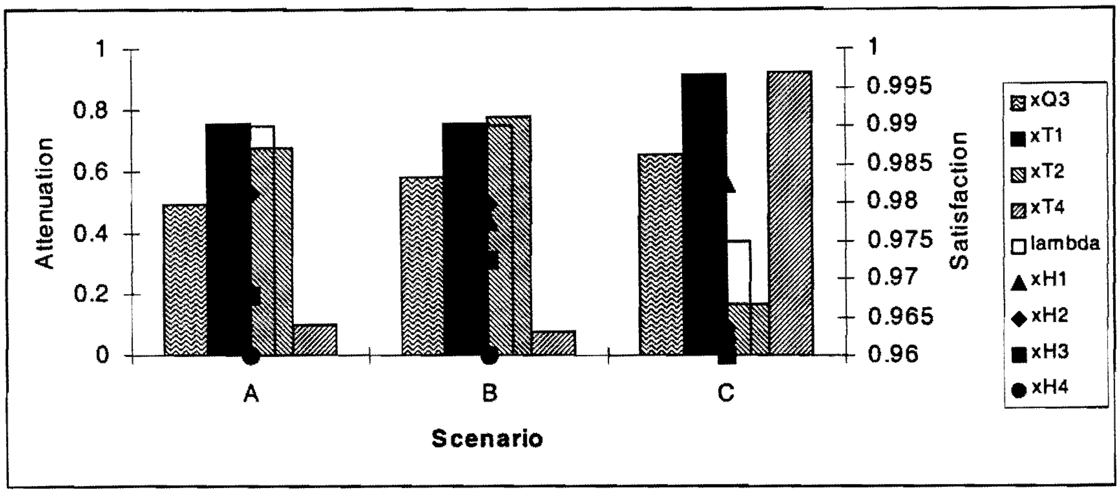

Figure 4. The effect of the removal of the equity constraint on the overall satisfaction λ and the attenuation levels (x) for flow (Q), toxics (T) and habitat (H) related stressors (for each of the 4 sources in the example) corresponding to the highest value of the overall acceptability λ. The value of λ is represented by the open rectangle and refers to the right hand ordinate axis.

If the equity constraint is removed (Fig. 4), much higher overall acceptability levels are reached. As expected, there is now also a much higher variability in stressor attenuation levels. Tightening the concern (risk) bounds highlights the more important contributors to ecological concern. In this case, habitat degradation downstream of node 4 with some contribution from toxics at node1, are probably the main contributions. It is interesting to note that flow is lower when the equity constraint is removed.

As could be expected from the values in Tables 1 and 2, toxic emission attenuation impacts the most on the ecological concern values and consequently demands the highest attenuation. However, while it would normally have been expected that sources 1 and 2 would require the highest attenuation (Figure 4), equity considerations lowers the attenuation for these sources at the cost of increasing attenuation at source 4 (Figure 3). The feasibility of doing this would obviously depend on local conditions. Although the true optimum may not have been reached on the imposition of equity constraints, it would seem likely that the abstraction attenuation would be higher compared to the situation where equity is not required. This would be the result of the greater weight accorded to the larger number of sources: a larger number of abstractors in the system would have evened out this effect. In the South African situation, for example, given the relative scarcity of water and the dependence of agriculture on irrigation, equity constraints may well have to be waived. This would clearly be a matter of negotiation or policy.

Other results (Jooste 2000) confirmed that λ_x tends to dominate the overall acceptability of the solutions and that λ_R and λ_{eq} tended to be much higher than λ_x. While ecological concern considerations would appear to raise the attenuation values, the source- and stressor specific acceptability consideration are still limiting. The implication here is that, unless the factors determining attenuation acceptability criteria are addressed, no further impact reduction could be expected. Since these factors may include both economic and technological considerations, addressing them may also have far reaching ramifications.

These results and the assumptions on which they were based would have definite policy and catchment management implications. However, the results in themselves may serve as a useful tool in decision making, supplying at least a baseline for decision making with a view to ecological protection.

CONCLUSION

Ecological concern, like ecological risk, makes use of available data on both the occurrence of stressors and the expected effect of these stressors. The likelihood nature of ecological concern lends itself to the aggregation of the contribution of diverse stressors if a common effect (such as loss of sustainability) is chosen as an end-point. However, it requires an explicit statement of at least semi-quantitative concern (or risk) objectives.

The ecological concern approach to stressor management may prove to be a useful tool in water resource management policy formulation as well as situation analysis under conditions where ecological goals need to be integrated with point source management issues. Although the information requirement for this approach is not insignificant, it provides a platform on which water quality and quantity issues can be integrated. It may be a basis on which stressor and source specific criteria can be generated. The practicality of this methodology would be influenced by a) the knowledge base available to estimate the conditional effect possibility, and b) the spirit of co-operation among the regulator and the stressor-source manager.

It is recognised that the estimation of the conditional effect possibility and the stressor attenuation acceptability criteria as described here, is essentially subjective. This process needs to be formalised and refined possibly drawing on the extensive work done on fuzzy expert systems. The formulation of objective procedures to derive these critical parameters will certainly facilitate the use of ecological concern as a water resource management tool.

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