# 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($\alpha$, $\beta$): | 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, overharvesting 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 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).

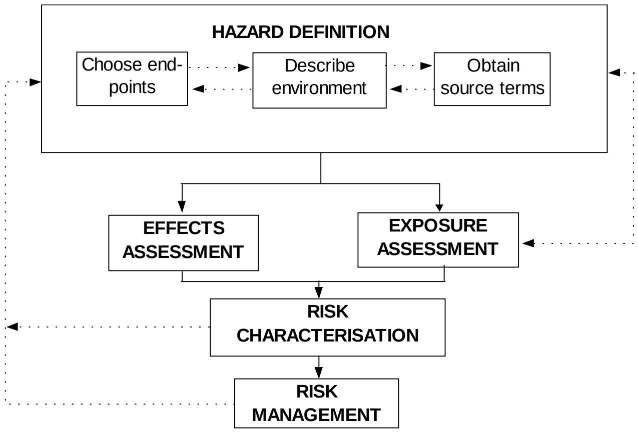


***Figure 1***
*A diagrammatic representation of the predictive use of ecological risk assessment (from Suter, 1993). The dashed lines indicate feedback loops.*

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):

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

It is recognised that $P(E_t)$, $P(E_q)$ and $P(E_h)$ are joint probabilities of effect $\varepsilon_x$ and exposure *x* so that: $P(E_x) = P(\varepsilon_x, x) = P(\varepsilon_x \mid x)P(x)$, 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).

$P(\varepsilon_x \mid x)$ 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 $\varepsilon_x$ 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 $P(\varepsilon_x \mid X)$ 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):

$$P(\varepsilon_x \mid X) = \sum_j (\mathrm{PX}_j) \cdot P(\varepsilon_x \mid X_j) \tag{4}$$

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 $P(\varepsilon \mid X_iX_j)$ would be given by $P(\varepsilon_Y \mid Y) = P(\varepsilon, Y) \mid P(Y)$. Therefore, $P(\varepsilon, X_i, X_j) = P(X_i)P(X_j)P(\varepsilon \mid Y)$, where the value for $P(\varepsilon \mid Y)$ 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<br>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.

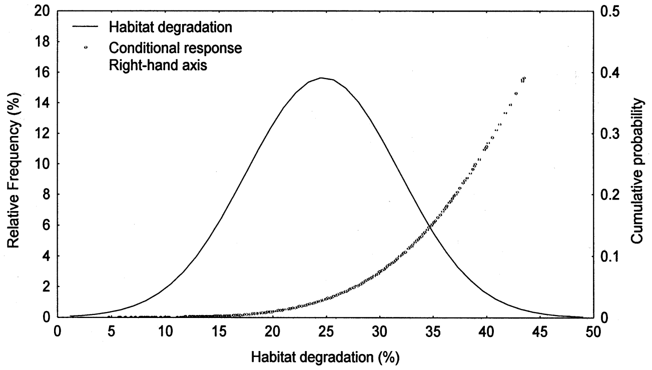


***Figure 2a***
*Habitat degradation distribution as used in the Monte Carlo simulation and the conditional probability of system response (points referring to the right-hand ordinate scale)*

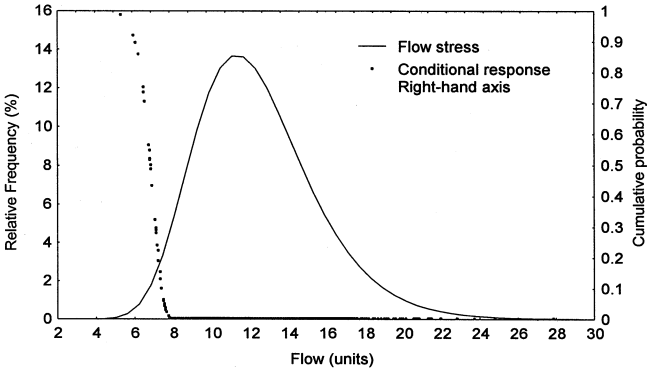


***Figure 2b***
*The flow-related stressor magnitude distribution (solid line) and the corresponding conditional system response probability (point referring to the right-hand ordinate scale)*

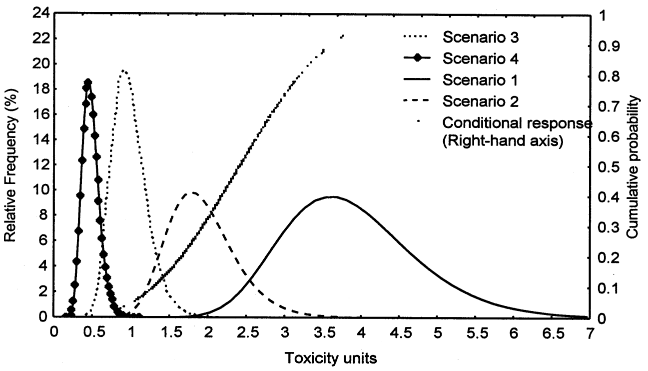


***Figure 2c***
*The toxic unit distribution for the four scenarios described in the text (lines referring to the left-hand ordinate scale) and the conditional system response profile for the toxic substances (the points referring to the right-hand ordinate scale)*

## 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-

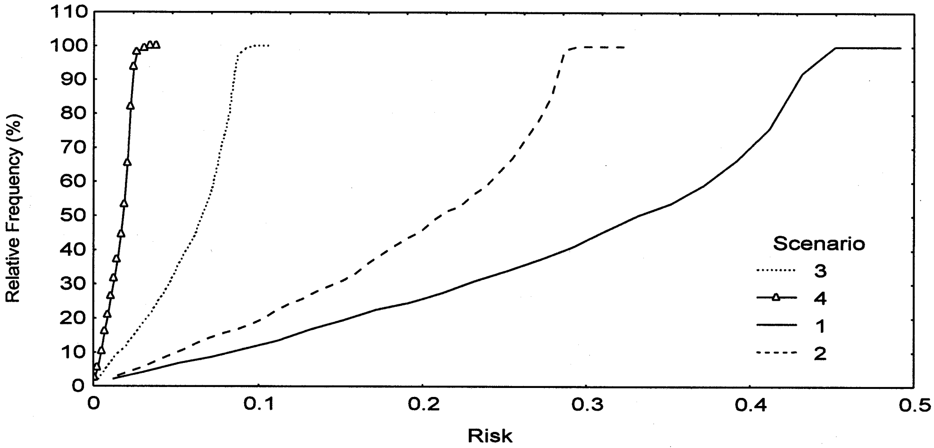


***Figure 3***
*The cumulative probability profiles of the overall risk from the Monte Carlo simulation of the scenarios described in the text*

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 (90$^{th}$ 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 90$^{th}$ 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 50$^{th}$ and 90$^{th}$ 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 ($\beta$) of the Weibull function would have the greatest uncertainty and that the uncertainty in $\beta$ can be described by a normal distribution. The increase in uncertainty is reflected in an increase in the relative standard deviation (RSD, ratio of

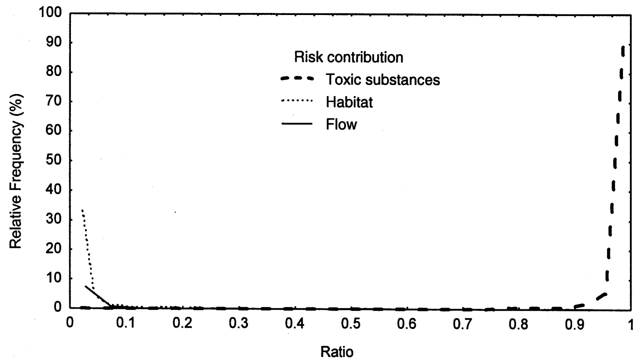


***Figure 4***
*The ratio of stressor-specific risk to the overall risk for Scenario 1*

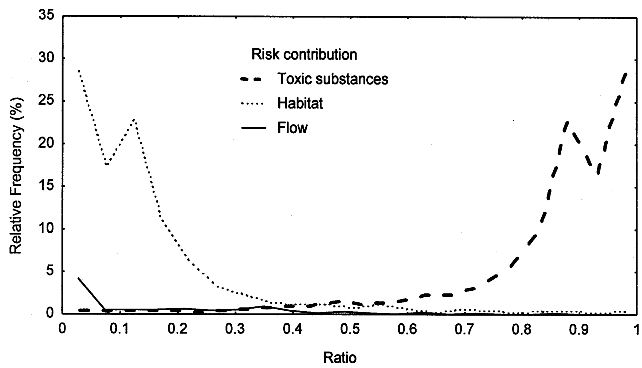


***Figure 5***
*The ratio of stressor-specific risk to the overall risk for Scenario 4*

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 50$^{th}$ and 95$^{th}$ 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 500$^{th}$ and 950$^{th}$ 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.

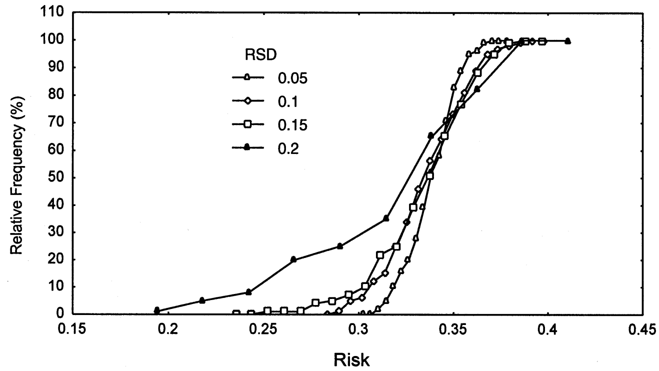


***Figure 6a***
*The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the median risk value*

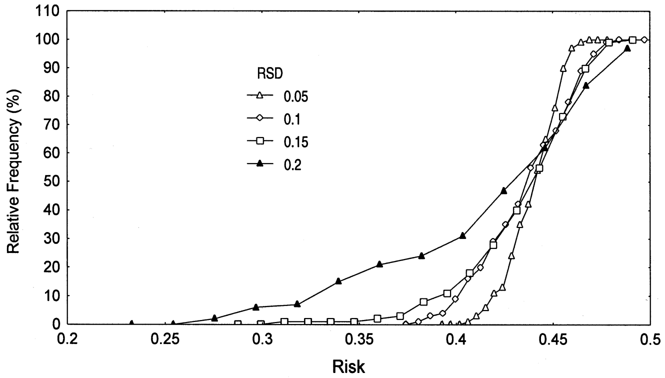


***Figure 6b***
*The effect of location parameter uncertainty (as reflected by the RSD) on the distribution of the 95th percentile risk value.*

## 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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