

# CATCHMENT-SCALE ECOLOGICAL RISK ASSESSMENT: A REVIEW

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## 1. INTRODUCTION

Agrochemicals are widely used in agricultural industries to protect or promote the growth of a crop. Although many chemicals are intended to target certain pest organisms, they are generally non-selective and as such they have the potential to affect other non-target organisms, sometimes inducing toxic responses, when habitats become contaminated. The impacts are commonly investigated through an ecological risk assessment (ERA) process.

Agrochemicals are referred to as chemicals applied in agricultural production. Such chemicals range in their mode of action, whether it is to support the growth of a crop e.g. fertilisers, or to control unwanted pests e.g. pesticides. Although their uses deliver strong benefits to agricultural production, their impacts on non-target organisms have also been widely documented. Subsequently, the fate and behaviour of chemicals in the environment is assessed to ascertain potential environmental contamination scenarios (Kookana *et al.* 1998). Although contamination may be anticipated, comprehensive assessments are required to assess the impact that exposure may have on ecological groups. It is for this reason that ERA's are carried out.

Ecological risk assessment (ERA) is a decision support framework designed to aid decision making under significant uncertainty (Suter 2007; USEPA 1998). The process evaluates the likelihood of adverse effects towards ecological groups resulting from exposure to one or more stressors. It has evolved to become one of the most commonly used frameworks to support decision making for the management of hazardous chemicals worldwide. The ERA framework was initially developed by the United States Government to characterise impacts that hazardous compounds have on humans, particularly in relation to carcinogenesis (Landis 2003). It was later modified to include ecological groups (Landis 2003; Suter 2007). The description of how ecological processes are affected by exposure to toxic compounds, combined with modelling and evaluation using comparative toxicological studies led to the development of ecological risk assessment (Landis 2003; Suter 2007). An ERA may also be used to support prioritization of pollutants or sites for regulatory purposes,

as well as in the development of environmental quality guidelines (Solomon and Takacs 2001). The most commonly adopted framework, that is subsequently optimised worldwide, was developed by the United States Environmental Protection Agency (USEPA) in 1998.

Management of natural resources around the world is increasingly being implemented at the catchment-scale. As water quality is dependent on processes occurring in the catchment it would be envisaged that the management of agrochemicals should also reflect natural resource management scales. Subsequently, this has instigated the modification of the ERA approach to operate at catchment-scales (Serveiss 2000). However, accounting for the complexity within catchment boundaries is often difficult to assess. This has resulted in the development of more advanced spatial approaches by applying ERA to geographical information systems (GIS).

The purpose of this review is to describe the extent that agrochemicals, particularly pesticides, are being used and managed in Australia, with special reference to the Australian cotton industry. The processes that determine the fate and persistence of agrochemicals in the environment with reference to the pesticides diuron and endosulfan will be reviewed. In addition, catchment-scale approaches to ecological risk assessment will also be reviewed and scope for applying catchment-scale ecological risk assessment to a GIS platform, with specific application to Australia, is also discussed.

## **2. SCOPE OF AGROCHEMICAL USE**

Pests, inclusive of weeds and insects, impose major constraints on production of many crops worldwide through direct yield reductions, damage in storage and costs associated with control measures (Fitt *et al.* 2004; Kookana *et al.* 1998). The control measures employed range from the use of organic and inorganic chemicals (herbicides and insecticides) to biological methods. The common means of pest control involves the use of synthetic organic chemicals. Their use dramatically increased after World War II because of their cost-effectiveness, ubiquity in application and efficacy for control (Kookana *et al.* 1998). Prior to the 1980's weed control was achieved through conventional tillage practices. In recent times there has been a large implementation of minimum tillage practices which are seen to benefit soil quality by reducing erosion and compaction, and increasing soil organic matter. The primary mechanism by which weeds were controlled was deemed bad practice and the dependence and demand for herbicides increased substantially (Kookana *et al.* 1998). The use of pesticides has become an integral part of pest management in agricultural production worldwide. Australian agricultural production is highly dependent on pesticide use as a

means of optimizing production and yielding high quality produce to meet local and international demands. This section reviews agrochemical use in agriculture, using the Australian cotton industry as an example.

The use of synthetic organic pesticides has contributed significantly towards the increase in global food and fibre production. It has been estimated that Australia alone has benefited from their use, in the range of \$4-5 billion per annum (Combella, 1989; as cited by Kookana *et al.*, 1998). However, even after pesticide application financial losses due to weeds, for example, is estimated to be about \$3.3 billion (Combella, 1989; as cited by Kookana *et al.*, 1998). Although the financial benefits are recognised as substantial, their impacts on the environment are also well known, notably toward aquatic ecosystems (Rose *et al.* 2005a; Rose *et al.* 2005b; Shivaramaiah *et al.* 2005).

As an example, the Australian cotton industry, which has generates over \$1.5 billion per year in export revenue (Cotton Australia 2006), has been recognised for its extensive use of agrochemicals that impact on the environment. Much of Australia's cotton is produced in NSW under irrigation, with water derived from head rivers of the Murray-Darling River catchment (Cotton Australia 2006; Raupach *et al.* 2001). The cotton industry was formerly recognised as one of the largest users of chemicals in Australian agriculture (Fitt *et al.* 2004; Kennedy *et al.* 2001; Kookana *et al.* 1998; Rose *et al.* 2005b) and has been implicated for serious environmental problems, notably in contamination of surface water bodies (Agassi *et al.* 1995; Kennedy *et al.* 2001; Muschal and Warne 2003) and more recently groundwater reservoirs (Accinelli *et al.* 2002; Landry *et al.* 2006). In 1985, the NSW State Pollution Control Commission (SPCC) recognised this potential, notably in cotton production areas associated with inland water bodies of North-Western NSW (Muschal and Warne 2003; Shivaramaiah *et al.* 2005). Such rivers commit to supplying water to irrigation regimes, livestock watering, and general domestic use. They are becoming increasingly designated for environmental flows and licensing schemes have been introduced to limit degradation and promote sustainable use of water resources (Muschal and Warne 2003). However, cotton farmers are required to recycle tail water, as water becomes increasingly viewed as a finite resource and to limit off-site movement of pesticide contaminated water. This requirement is contained within the implementation of best management practices (BMP) (Cotton Australia 2006).

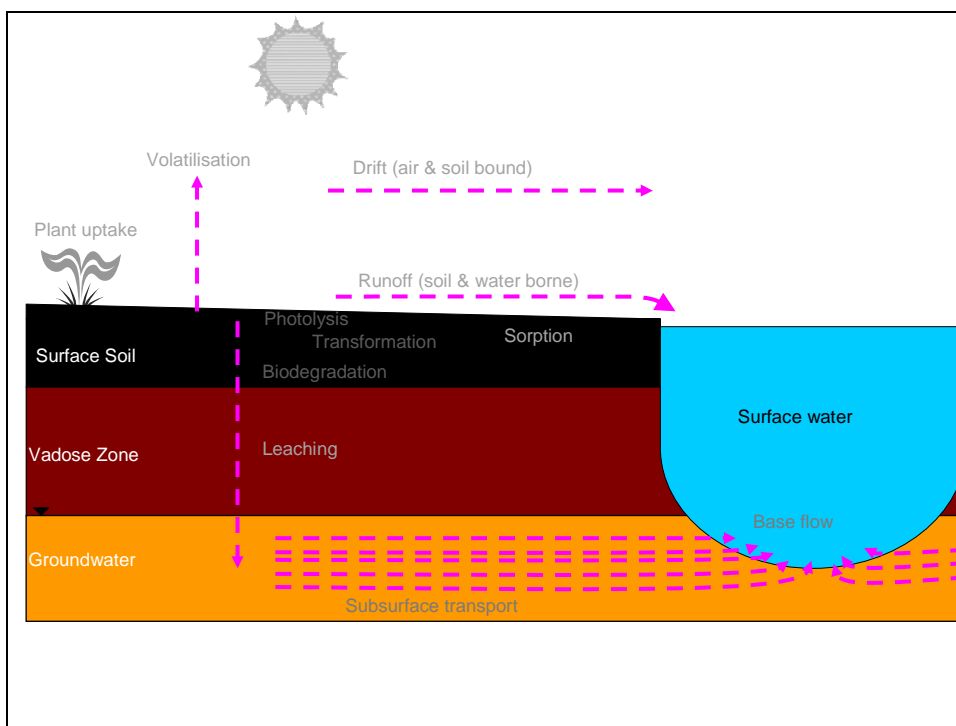
An increasing view has been adopted by the wider international community that agricultural practices must exhibit minimal environmental impacts for enhanced market vigour (Muschal and Warne 2003). This has provided the incentive that governments and

industry to practice environmental conservation in order to protect lucrative export markets by virtue of maintaining a 'clean and green' image in practice while retaining profitability (Muschal and Warne 2003; Nett and Hendley 2002). For example, the Australian cotton industry has developed several strategies aimed at reducing the amount of agrochemicals used in production as well as minimising environmental impact (Kennedy *et al.* 2001). Such strategies include Best Management Practices (BMP), Integrated Pest Management (IPM) (Fitt *et al.* 2004; Yee and Ferguson 1996), and recognition and ratification of international standards (such as ISO). In meeting these strategic goals, significant research has gone into the development of new methods that reduce the use and overall impact of pesticides. Such technologies include genetically modified (GM) insect-resistant cotton (e.g. Ingard and Bollgard<sup>®</sup>), construction of on-farm holding dams to collect contaminated runoff (Cotton Australia 2006; Kennedy *et al.* 2001), and bioremediation through constructed wetlands and subsurface filtration (Rose *et al.* 2005a; Rose *et al.* 2005b). All of these technologies have been developed to reduce pesticide use and overall environmental impact to meet devised strategies and legislations.

Agrochemicals are adopted in agricultural production to control problem pests. This approach has been adopted to enhance the quality of yields. The outcomes of agrochemical use have been reported have significant environmental implications recognised at the international level. This has led to the development of technologies to limit environmental impacts and enhance crop vigour in international markets.

### **3. ENVIRONMENTAL PERSISTENCE AND FATE OF AGROCHEMICALS**

As mentioned earlier, agrochemicals are widely recognised to pose environmental hazard to non-target organisms when used in agricultural practices. Once applied to the target site, the fate and persistence of agrochemicals in the environment is controlled by biotic and abiotic processes illustrated in Figure 1. Such processes control the extent to which a chemical may be exposed to sensitive organisms. The extent to which a chemical can persist in a defined environment is quantitatively defined by the half-life of a compound. This section describes the different mechanisms that contribute to the persistence and fate of agrochemicals following their application in agricultural practices using the herbicide diuron and insecticide endosulfan as case study agrochemicals.

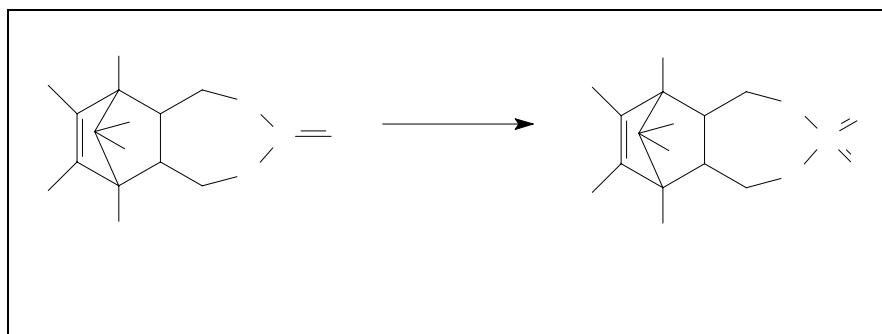


**Figure 1.** Various processes determining the fate of pesticides in the environment.

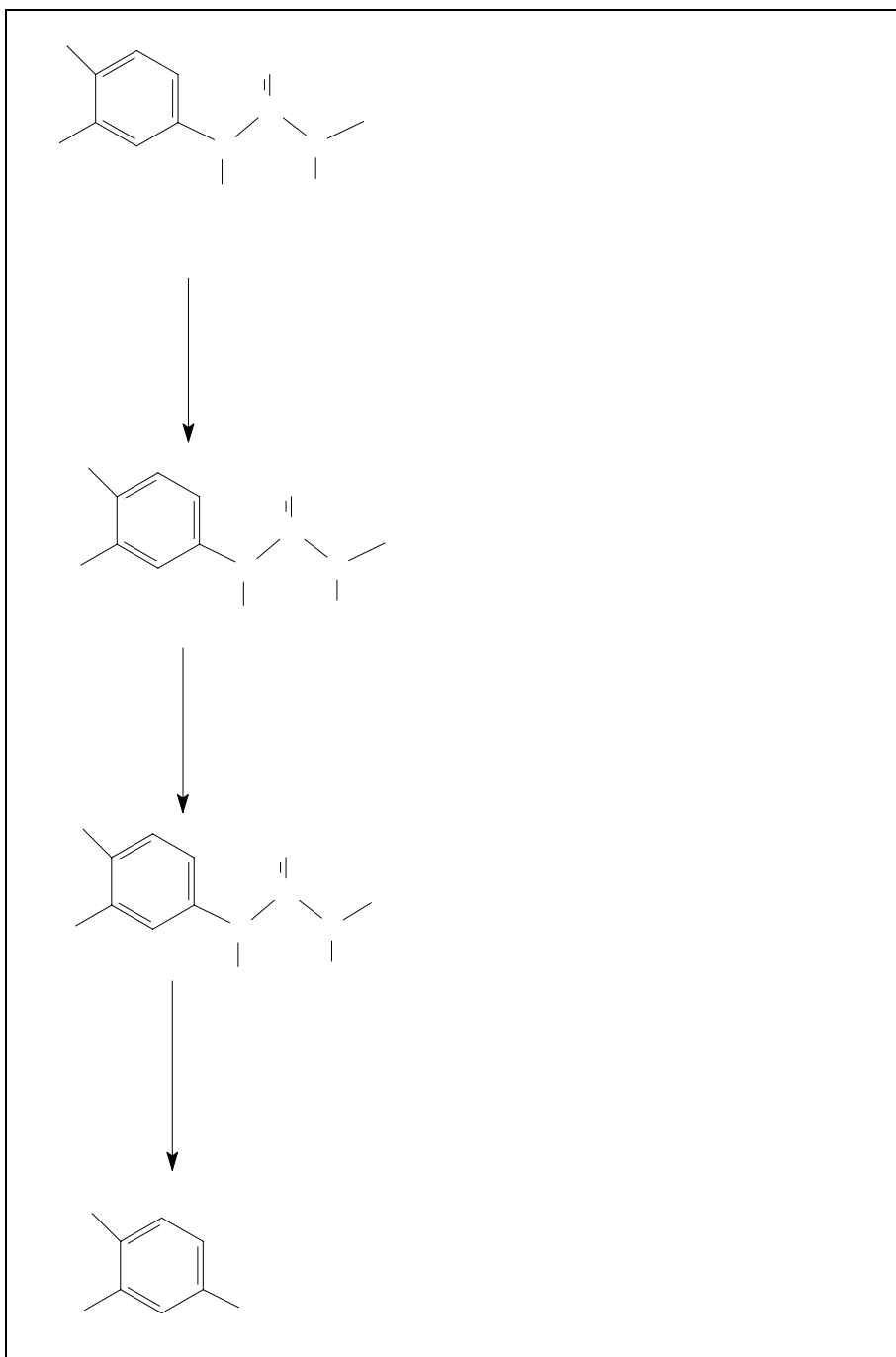
The intended targets of agrochemicals are weeds and insects. Application of the chemicals involves deposition on the crop foliage and soil. Accounting to its properties, the deposited agrochemical then undergoes volatilisation, degradation or is washed into the soil by rainfall or irrigation water, leached into the soil profile, or incorporated into the soil with crop residue (Wauchope *et al.* 2002). Agrochemicals also partition between the liquid, solid and gaseous phases of the environment, the relative distributions of which are described as partition coefficients (Ding *et al.* 2002; Schnoor 1992; Wauchope *et al.* 2002). Partitioning results from a series of complex interactions involving equilibrium tending to exist for chemicals between the soil, air and water phases of the environment (Ding *et al.* 2002). The extent to which a chemical partitions between these phases is influenced by the physicochemical interaction of the agrochemical and the harbouring medium. Sorption to soil is reversible, as desorption acts concomitantly with hysteresis sometimes being observed for irreversibly bound compounds. Such processes provide an indication of bioavailability of agrochemicals (Ding *et al.* 2002; Schnoor 1992; Wauchope *et al.* 2002). The prevailing component that enhances the sorption of largely hydrophobic agrochemicals to solid phases is organic carbon content of the sorbing media. This has been shown for many agrochemicals, for example the herbicide diuron (3-(3,4-dichlorophenyl)-1,1-dimethylurea) (Gaillardon 1997; Gooddy *et al.* 2002; Gouy *et al.* 1999; Landry *et al.* 2006; Liyanage *et al.* 2006; Roy *et*

*al.* 2000; Sheng *et al.* 2005; Yang and Sheng 2003a; b) and the insecticide endosulfan ((1,4,5,6,7,7-hexachloro-8,9,10-trinorborn-5-en-2,3-ylenebismethylene) sulfite) (Ghadiri and Rose 2001; Kumar and Philip 2006; Marshall and Rutherford 2002).

As well as its partitioning, the persistence of an agrochemical at the point of application is governed by both biotic and other abiotic processes. Biotic mechanisms in soil include plant uptake and microbial degradation; and abiotic processes include chemical and photochemical degradation (Kookana *et al.* 1998). The processes may result in mineralisation by microbial metabolism, photochemical degradation, or surface-catalysed reaction into inorganic compounds such as carbon dioxide, or ammonia (Baer and Calvet 1999; Jacobson *et al.* 2005). The processes result in the transformation of the parent compound to form biproducts that are chemically and physically different. For example, oxidation of the insecticide endosulfan readily occurs in water to produce endosulfan sulfate (Figure 2; Kennedy *et al.* 2001). Endosulfan sulfate is considered to be slightly less toxic than its parent isomers; however it is more soluble and persistent than the parent isomers (Kennedy *et al.* 2001; Leonard *et al.* 2001; Peterson and Batley 1993). Microbial degradation is considered to be the primary mechanism for diuron dissipation from soil (Gooddy *et al.* 2002). Under well-oxygenated conditions aerobic degradation, by way of oxidation, has been reported to result in successive demethylation of its urea group, followed by hydrolysis, to ultimately yield 3,4-dichlorophenylaniline (Figure 3; Gooddy *et al.* 2002). As the transformation products may be more or less toxic to organisms, they should be considered when assessing exposure toxicity.



**Figure 2.** Oxidation of endosulfan to endosulfan sulfate (after Kennedy *et al.*, 2001).



**Figure 3.** Oxidative demethylation pathway of diuron (after Gooddy *et al.*, 2002)

At the same time as undergoing partitioning and degradation processes, agrochemicals are also subject to transport processes. The extent to which a chemical is mobilised in the environment is mostly dependent on its ability to partition between the different environmental phases, solid, liquid and gas; and proximity of application to weather events. Agrochemicals may undergo volatilisation and subsequently drift through the atmosphere, leach through the soil profile as a soluble load to aquifers and be entrained in

runoff as a soluble load or sorbed to sediment. For example, endosulfan is considered very volatile and hydrophobic, as respectively indicated by its very high vapour pressure (0.83 mPa for a 2:1 mixture of  $\alpha$ - and  $\beta$ - isomers at 20 °C; (Tomlin 1997) and low solubility in water ( $\alpha$ -endosulfan = 0.32 mg L<sup>-1</sup> at 22 °C, and  $\beta$ -endosulfan = 0.33 mg L<sup>-1</sup> at 22 °C; Tomlin 1997). It is for these reasons that endosulfan has been observed to reside strongly on the solid phase as well as volatilise and be transported as drift. Alternatively, diuron is considered very mobile as both sorbed to entrained sediment, and its slight polarity attributes to its higher solubility in water (36.4 mg L<sup>-1</sup>; Tomlin 1997) allowing it to be transported in the soluble load in runoff and leachate in infiltrating water. Although agrochemicals may be transported in the different phases that constitute the environment, ecological exposure concern is generally directed at the phases which are considered bioavailable, such as the gaseous and liquid phases, with the solid phase often considered as biologically unavailable.

The major concern for contamination occurs during the time of application. For example, drift concerns arise during the application of agrochemicals resulting from differing application methods. Agrochemicals may be applied directly using a boom spray or aerial crop dusting. If the dispensed droplets are fine enough they may be entrained in the atmosphere and transported offsite, pending the prevailing wind conditions, and contaminate non-target ecosystems. Further to this, deposited chemicals are also vulnerable to being transported in runoff if rainfall events occur immediately following application. Subsequently, restrictions have been imposed on the way and environmental conditions which agrochemicals may be applied to the target surface. For example, the recommended application rate of endosulfan in cotton production is 2.1 L ha<sup>-1</sup> of active ingredient and is recommended to coincide with egg-hatching of caterpillars or at the first sign of infestation (Farrell and Johnson 2005). Application, however, is limited to a total of 2205 g ha<sup>-1</sup> of active ingredient application per season, where irrigation tailwater and up to 25 mm of rainfall can be captured on farm (Farrell and Johnson 2005). This is reduced to 1470 g ha<sup>-1</sup> of active ingredient where irrigation tailwater and up to 25 mm of rainfall cannot be captured on farm (Farrell and Johnson 2005). The application of endosulfan has reduced compared to the 1990's and is restricted where drift may be prominent, i.e. high winds, and the subsequent mode and windows of application is also restricted in certain regions during the growing season, as Figure 4 reveals.



Conditions of Use on Cotton							
October		November		December		January	
1st-18th	18th-31st	1st-14th	15th-30th	1st-14th	15th-31st	1st-15th	15th-31st
Endosulfan EC – Ground Application							
		Endosulfan EC – Aerial application					
		Endosulfan EC – Aerial application					
All cotton growing areas except for the areas listed below. Shire council areas Bourke and Walgett in NSW, and Balonne, Banana, Bauhinia, Belyando, Broadsound, Dalrymple, Duaringa, Emerald, Peak Downs, Richmond and Waroo in QLD. DO NOT USE BY AERIAL SPRAYING ON COTTON LESS THAN 20 CENTIMETRES IN HEIGHT							

**Figure 4.** In season mode of application recommendations for endosulfan (from Farrell & Johnson, 2005).

The degradation and transport processes may result in the production of more potent and persistent compounds and transference of toxicity to other unrelated areas respectively. Degradation can be affected by many factors including changes in the physicochemical properties of the chemical and management of the soil through cultivation practices or fallow periods. Additionally, bioavailability of a compound can be reduced because of complexation or sorption to soil solids. The ability of microbes to degrade a compound depends on its bioavailability, which in turn affects its longevity (Jacobson *et al.* 2005; Scheunert 1993; Schnoor 1992). The literature reflects this by reporting half-life ranges; for example, diuron is considered to be moderately to highly persistent in soil with first-order half-lives ranging from 1 month to 1 year (EXTOXNET 1996), and endosulfan is moderately persistent in the soil environment with a reported average half-life of 50 days in the field. It is important to note that these half-lives have been derived from soils of different geographical locations, indicating that persistence is dictated by the environmental conditions exhibited by a site; for example temperature, moisture conditions, etc (Kookana *et al.* 1998; Nkedi-Kizza *et al.* 2006; Schnoor 1992).

It has been shown that there are a number of degradation and transport processes that influence the persistence and fate of agrochemicals in the environment. Such processes play determining roles in the exposure to non-target organisms. It was further revealed that the application regimes of certain chemicals have been devised based on prevailing conditions

that may influence the transportability of agrochemicals. As agrochemicals have the potential to be transported from the application target, contamination of non-target ecosystems is prominent and warrants further assessment.

#### **4. NATURAL RESOURCE MANAGEMENT IN AUSTRALIA**

Agrochemicals are widely used in agricultural industries and their potential impacts on ecological systems are well characterised. The management of natural resources in Australia, with particular reference to water, occurs at the physical boundaries that control its movement, the catchment. The use of agrochemicals and the regulations that are imposed on them does not reflect this natural resource management scale and is often limited to the farm gate with little scope for spatial variability in exposure potential. This section discusses “catchment management” as a means of managing natural resources in the context of agrochemical use.

Catchments are often perceived as hydrologically isolated ecosystems and are therefore a logical environmental management unit (Serveiss *et al.* 2000). Catchment-scale management of natural resources is a philosophy that is being adopted worldwide as a means of managing environmental quality, especially with regards to water. In particular, managing the attributes that influence the quality of water flowing through a catchment has been deemed paramount. This has often evolved from recognition of the lead-on effects that result from poor quality water, such as impacts on ecosystems, suitable for human consumption and use in agricultural production.

The management of natural resources in Australia is increasingly being applied at the catchment-scale. For example, the state governments of Australia have established catchment management authorities (CMA) that manage catchments for both human and ecological purposes and work on incentives to promote adoption of best practices to manage the quality of natural resources, such as soil, water, etc. The issues that are most often dealt with are land salinity and water quality which is limited to general quality attributes, e.g. pH, electrical conductivity, nutrient concentrations, etc. However, the management of agrochemicals is beyond their scope and is the responsibility of other government departments, such as environment protection authorities.

Often when problems do arise in catchments blame is directed at the most likely source, such as farms, and the event not considered for ecological significance. The monitoring of natural resources in close proximity to areas where agrochemicals are used in Australia is limited and is an issue that requires further development, especially at the

catchment-scale. USEPA (1998) outlines a procedural framework, called ecological risk assessment, designed to overcome ambiguity of stressor contamination and its overall ecological significance (Solomon and Sibley 2002). Such a framework has been established for general use with stressors such as agrochemicals and has been further developed to operate at the catchment-scale (Serveiss 2002; Serveiss *et al.* 2000; Serveiss and Ohlson 2007), the details of which will be discussed in the proceeding section. Subsequently, there is scope for the development of a catchment-scale ERA framework to support agrochemical management in Australia (Hart *et al.* 2006).

The management of natural resources in Australia does occur at the catchment-scale. However, it appears that the management of agrochemicals occurs only up to the farm gate oblivious to a farms locality or catchment. There is a need to assess the ecological impact that agrochemicals have and one of the means this can be achieved is through the use and development of a rigorous catchment-scale ecological risk assessment framework, similar to that used by the USEPA.

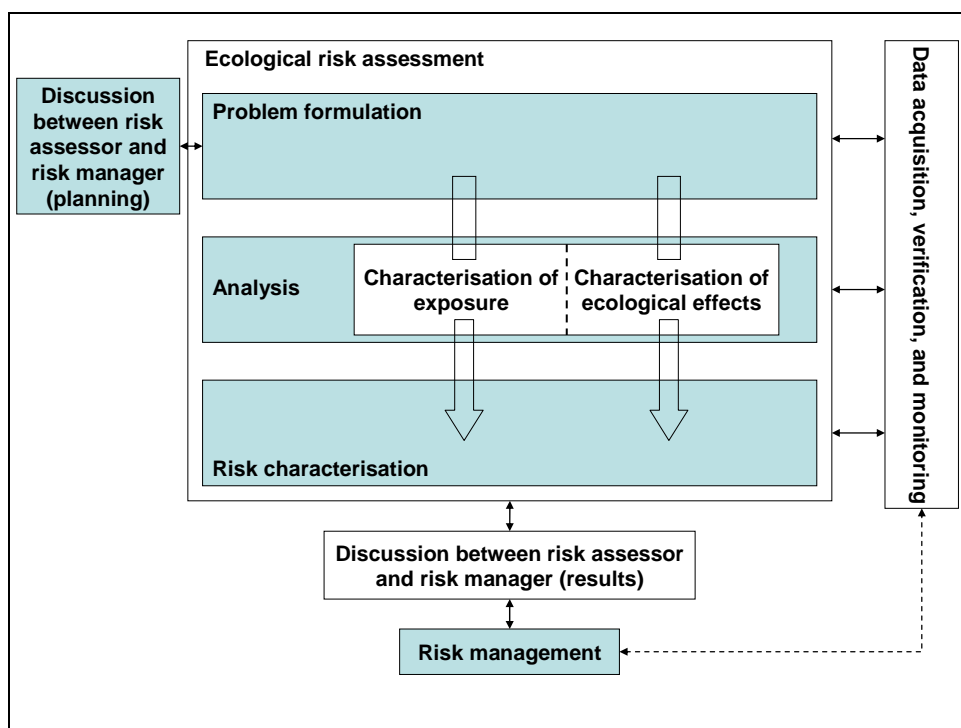
## **5. CATCHMENT-SCALE ECOLOGICAL RISK ASSESSMENT**

Following transport and degradation processes, ecosystems may become exposed to agrochemicals. The magnitude of an exposure and the potential for this to cause a problem is investigated. The common means which environmental management organisations, such as the USEPA, assess the impact that agrochemicals have on ecosystems is through an ERA. Ecological risk assessment (ERA) is a decision support framework designed to aid decision making under immense uncertainty (Suter 2007; USEPA 1998). The process evaluates the likelihood of adverse effects towards ecological groups resulting from exposure to one or more stressors. It has evolved to become one of the most commonly used frameworks to support decision making in the management of hazardous chemicals worldwide (Suter 2007).

The ERA framework was initially developed by the United States Government to characterise impacts that hazardous compounds have on humans, particularly in relation to carcinogenesis (Landis 2003). It was later modified to include ecological groups (Landis 2003; Suter 2007). The description of how ecological processes are affected by exposure to toxic compounds, combined with modelling and evaluation using comparative toxicological studies led to the development of ecological risk assessment (Landis 2003; Suter 2007).

The goal of ERA is to estimate the likelihood that adverse effects are an outcome from some level of exposure to a hazardous substance (Solomon and Sibley 2002; Verdonck *et al.* 2003). ERA follows a procedural framework that evolved from the National Research

Council framework for human health risk assessment (NRC 1983; as cited in Suter 2007). The framework serves to provide a basis for quality assurance by ensuring the necessary components are included. The most commonly adopted ERA framework that is subsequently optimised worldwide, was developed by the USEPA. The USEPA (1998) framework (Figure 5) is conceptually a tiered process involving three distinct steps of problem formulation; analysis (that involves characterisation of exposure and ecological effects); and characterisation of risk. Prior to the ERA a planning phase, involving discussion between risk assessors and risk managers is conducted; and a risk management phase follows the ERA (Figure 5). Throughout the ERA process new data may be acquired, verified and monitoring programs may be developed that may allow for optimisation of results accordingly. This section describes the catchment-scale ecological risk assessment framework used to evaluate agrochemicals.



**Figure 5.** USEPA ecological risk assessment framework (reproduced from USEPA 1998).

### 5.1 Planning phase

Planning is considered outside the scope of an ERA (Figure 5). Planning involves scientists (primary assessors) and managers (primary clients) together with stakeholders (Table 1) to come together and discuss the focus, scope and complexity of an ERA to be conducted (Cormier *et al.* 2000; Serveiss *et al.* 2000; USEPA 1998). The discussion highlights the need

for an ERA; the expected outputs from the assessment; indicate manager's intentions for using the information; identify technical and financial requirements to carry out the ERA; and highlight the benefits and limitations of the ERA process. Discussions should specifically deliver agreements on management goals, objectives and development of assessor's knowledge for the area being investigated. In essence, this stage assesses stakeholder concerns, potentially broadening the scope for investigation.

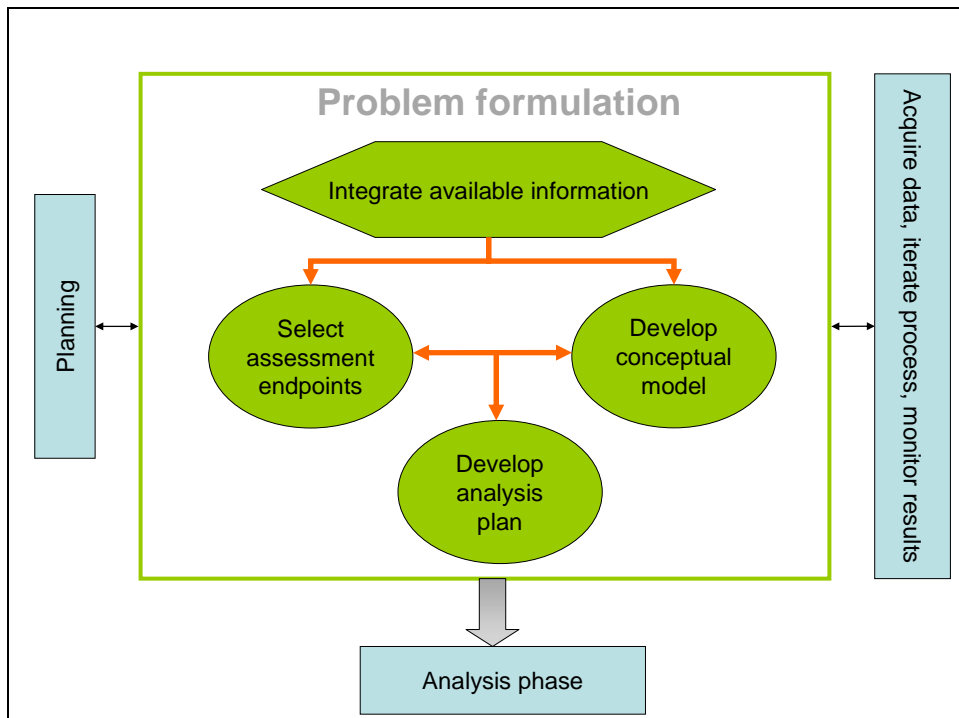
Catchment managers agree on management goals that will provide focus for the ERA. It may be that existing goals from community groups (e.g. landcare), local councils, catchment management authorities (CMA), state environmental management organisations (e.g. Queensland Environmental Protection Agency, New South Wales Department of Water and Energy, etc) and federal authorities (e.g. Australian Pesticide and Veterinary Medicines Authority (APVMA), Department of Environment, Water, Heritage and the Arts (DEWHA) and Department of Agriculture, Fisheries and Forestry (DAFF)) be used as a guide. Meetings with management groups provide scope for the development of shared management goals (Cormier *et al.* 2000). The management goals are generally supported by a set of more tangible management objectives that may be devised by a subgroup of the planning team (Cormier *et al.* 2000; Serveiss *et al.* 2000; USEPA 1998). Overall, it is the catchment goals and objectives that set the foundation for which the ERA is grounded.

**Table 1.** Summary of management groups to be included in discussions to support the ERA process (prepared by the Author).

Organisation level	Groups for inclusion
Local	Local government (councils) Other community action groups Private business
Catchment	Catchment management authority Landcare
State	New South Wales Department of Water and energy Queensland Environmental Protection Agency Environmental Protection Authority of Victoria Environmental Protection Authority of South Australia Department of Environment, Parks, Heritage, and the Arts Western Australian Department of Environment and Conservation Northern Territory Natural Resources, Environment, The Arts and Sport Australian Capital Territory Department of Territory and Municipal Services State research organisations
Federal	APVMA DEWHA Chemical companies Research organisations (e.g. CSIRO, Universities, etc)

### 5.2.1 Step 1: Problem formulation

Problem formulation is the phase that develops the organizing framework for the ERA (Figure 6) (USEPA 1998). The purpose of the phase is to summarise the complex environmental concerns, impacts and relationships observed in the catchment, and formulate preliminary hypotheses about why ecological effects have or may occur as a result of human activities (Cormier *et al.* 2000; Suter 2007; USEPA 1998). It makes use of available information that characterise ecological resources potentially at risk, stressors, and observed or anticipated ecological effects, to describe the nature of the problem and identify measurable attributes that can be used as measures of exposure and effect, and assessment endpoints. A conceptual model is then constructed to describe the interrelationships among resources, stressors and effects, and provide a focus for the assessment and a plan for the analysis phase of the ERA. Overall, problem formulation integrates available catchment information to aid in the selection of assessment endpoints, development of a conceptual model, and analysis plan to support the ERA (Figure 6) (Cormier *et al.* 2000; Serveiss *et al.* 2000; Suter 2007; USEPA 1998). The outcome is a clear focus for the ERA and a plan for the analysis phase (Phase 2).



**Figure 6.** Conceptual diagram of the problem formulation phase in catchment-scale ERA (taken from Serveiss 2000).

### *5.2.1 Gathering available information*

The first step in problem formulation is to gather information that enables the assessor to identify and characterise the ecosystem(s) potentially at risk, ecological effects and the stressors that are likely to contribute to the effects (Cormier *et al.* 2000; Serveiss *et al.* 2000; Solomon *et al.* 1996). Information may be sourced from published material such as government documents or scientific journals; and unpublished material, such as community websites or community meetings. The information enables identification of known and unknown relationships among stressors, exposure scenarios and effects that ultimately supplements the understanding of relationships that exist between catchment processes and risk (Cormier *et al.* 2000; Serveiss *et al.* 2000; Solomon *et al.* 1996). Importantly, the information also allows for the identification of information gaps to be included for further investigation. Prior to its use, the information is evaluated for validity to minimise uncertainty.

Available information was sought from a number of sources in ERAs carried out by Solomon *et al.* (1996); Cormier *et al.* (2000); Salihoglu and Karaer (2004); and USEPA (2002a; b). These included conducting comprehensive literature reviews, making use of unpublished information where possible, and conducting meetings with community and government groups. The information enabled for the development of an initial description of their respective catchments and highlight ecological resources at risk, as well as the development of a strategic plan that formulated the scope of the assessments. Management goals were developed by the assessment groups so that the ERA would address regulatory requirements and public concerns. The information was then used to support later steps in the problem formulation phase, such as identifying ecological resources at risk in the catchment, characterising stressors, selecting endpoints, and development of conceptual model(s); as well as later phases of the ERA.

### *5.2.2 Ecological resources potentially at risk*

While conducting discussions, ecological resources to be protected are evaluated. Such ecological group(s) exhibit value to the various stakeholders. Characterisation of such resources entails descriptions of the catchments ecosystems and provides the basis for evaluating concerns expressed by the stakeholders and the resources that may be at risk (Serveiss *et al.* 2000; USEPA 1998). The concerns are scientifically evaluated and include how and where adverse effects are likely to be observed in the area.



Characterisation of ecological resources at risk often entails a description of the catchment or ecosystem to be investigated. Cormier *et al.* (2000); Hall (1998); Hall *et al.* (2000); Schuler and Rand (2008); USEPA (USEPA 2002a; b) identified ecological resources that had potential risk using supporting information, such as monitoring programs. Another means of identifying resources at risk is to use environmental fate principles, described earlier, to characterise stressor exposure scenarios based on spatial characteristics of chemical use, such as that described by Solomon *et al.* (1996) for atrazine exposure in freshwater systems of the US Midwest. Assessing environmental fate principles and constructing a conceptual model that describes the fate and transport processes likely to influence the exposure of agrochemicals in question is also useful in evaluating ecosystems at greatest risk.

### 5.2.3 Stressor characteristics

A stressor is any physical, chemical or biological entity that exerts an adverse effect to a non-target organism. A range of stressors can exert some form of effect in a catchment. Stressor sources include human activities and natural processes occurring in the catchment. It is important to characterise the stressor in terms of type (chemical, physical, biological), intensity (concentration or magnitude), duration (short or long term), frequency (one time, episodic, continuous), timing (relative to seasons, life cycles) and scale (extent, spatial heterogeneity) of exposure (Serveiss *et al.* 2000; Suter 2007; USEPA 1998). To supplement characterisation of the stressor(s), information on ecological effects (such as EC50s or LC50s) is often gathered to anticipate how stressor(s) pose specific risks in a watershed.

Initially, a comprehensive description of the stressor is carried out. This involves describing the chemical and physical nature of the chemical, its intended use and associated mode-of-action and recommended application regimes as well as its behaviour in the environment. Important physical and chemical attributes include solubility in water; vapour pressure (ability to volatilise); fugacity; partition between solid, liquid and gas phases; a range of possible environmental half-lives; and possible breakdown products. Organisms likely to be affected are also described, for example toxicity information (EC50s, LC50, etc.) may be used as a guide.

Cormier *et al.* (2000); Salihoglu and Karaer (2004); USEPA (2002a; b); and Solomon (1996) provided a comprehensive characterisation of their stressors to be investigated. Stressors included sediment and nutrients (Cormier *et al.* 2000; Salihoglu and Karaer 2004; USEPA 2002a; b), atrazine (Solomon *et al.* 1996), and other agrochemicals (Schuler and Rand 2008; USEPA 2002a; b). They characterised their stressors in terms of scenarios that

were likely to result in exposure risk accounting for type, intensity, duration, frequency, timing and scale of exposure. Some characterisations went further to account for a possible range of effects using among species tolerance distributions, population and community responses (e.g. microcosms/mesocosms used by Solomon *et al.* 1996), mode-of-action, interactions of duration and intensity of exposure on effects and recovery rates (Schuler and Rand 2008; Solomon *et al.* 1996; USEPA 2002a; b). Spatial distributions of stressors were accounted for where possible, for example Solomon *et al.* (1996) characterised the distribution of atrazine use in Mid West region of the US in a map. Subsequently, adequate stressor characterisation provided key guidance on where chemicals are likely to pose risks in the area and identify areas for further investigation.

The available information provided on ecological resources, stressors and effects are then used to formulate assessment endpoints that involve identifying and selecting the specific subjects of the assessment; develop a conceptual model and questions that the assessment may address; and develop a plan of action for the analysis phase and necessary measurements.

#### 5.2.4 Endpoint selection

Endpoints are an expression of the environmental value to be protected (Suter 1990; Suter 2007; USEPA 1998). They provide the link between what can be measured (e.g. fish species richness) and management objective(s) (e.g. protecting one or more fish species). In essence, they provide explicit definitions of a clear focus for the risk assessment (Serveiss *et al.* 2000; Suter 2007; USEPA 1998).

Two types of endpoints are used in ERA, assessment endpoints and measurement endpoints (Suter 1990; Suter 2007). Assessment endpoints are expressions of environmental values that are to be protected. A measurement endpoint is an expression of an observed or measured response, generally of an environmental characteristic related to the valued attribute chosen as the assessment endpoint, to the hazard (Solomon *et al.* 1996; Suter 1990). The selection of assessment endpoints is critical as they translate intangible management goals to scientific measurements. Conceptually, the endpoints relate to the management objectives and key characteristics of the valued ecological resource identified during planning.

The formulation of endpoints involves recognising the dynamic, complexity, and deterministic and stochastic components of an ecosystem (Leuven and Poudevigne 2002). Three criteria are provided by USEPA (1998) and Serveiss (2000) used in selecting

assessment endpoint, these include (1) relevance to important traits of the ecological resource at risk; (2) relationship to policy goals and resources valued by the community; and (3) susceptibility to the stressor. Further considerations for large scale ERA endpoints have been described by Suter (1990). Although there is much difficulty in measuring assessment endpoints (Serveiss *et al.* 2000; Suter 1990) where direct measurement may not be possible, an alternative means is to select measures of effect (or measurable responses to a stressor) (Serveiss *et al.* 2000). They are chosen on the grounds that they are suitable for detecting changes to the broader assessment endpoint, singly or in groups, as well as for their ability to be accurately, consistently and economically measured.

Assessment endpoints were devised by Cormier *et al.* (2000); Salihoglu and Karaer (2004); USEPA (2002a; b); and Solomon (1996). The endpoints were selected with considerations of management objectives, with explicit reference to what was to be protected. The endpoints were clear statements of what was intended to be achieved. For example, USEPA (2002a) devised two endpoints that stated the assessment intentions of accounting for “Diversity and abundance of threatened, endangered, or rare native freshwater [(endpoint 1)] mussels [and (endpoint 2) fish] species.” Although these endpoints focus on a single organism, an endpoint may focus on the integrity of a group of organisms (e.g. Solomon *et al.* 1996). Following these statements, an assessment of the endpoints importance was prescribed to provide justification for its selection. The risks posed to the endpoint by linking them with the stressors likely effects following exposure and organisms relationship with the catchment (including distribution and behaviour) under defined conditions. In essence, the management goals were defined to provide a clear focus for the assessment.

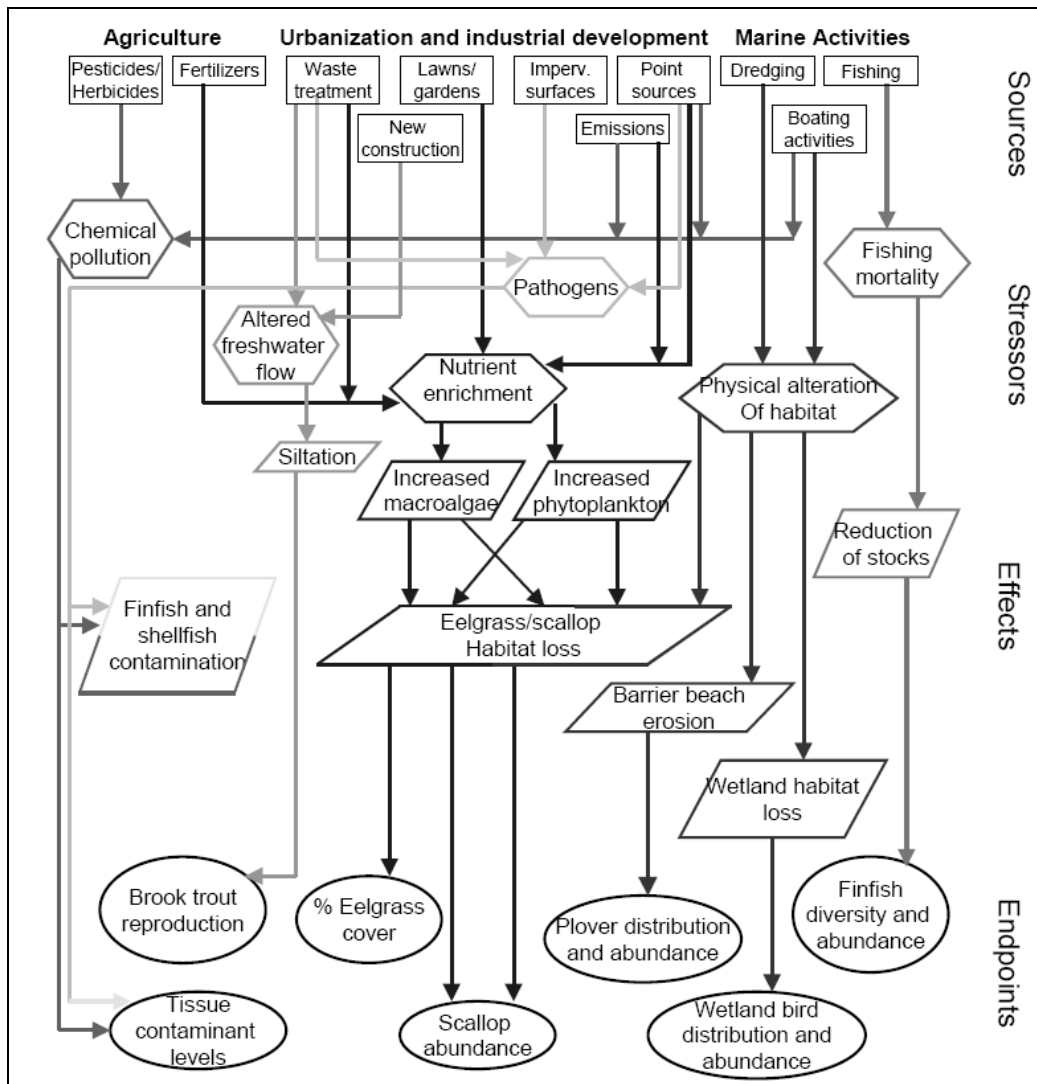
#### 5.2.5 Conceptual model development

The aim of a conceptual model is to utilise the information devised in the preceding steps. The conceptual model draws on relationships among sources of stress, stressors, ecological effects and endpoints. A framework for the analysis and assessment of information is also described. An analysis plan prescribing the exposure/effects relationships that are to be quantified in the analysis phase, the data requirements and measures to be used, and how the risk will be described (Cormier *et al.* 2000; Serveiss *et al.* 2000; USEPA 1998), is also presented. The conceptual model is presented in both narrative and graphical form.

Conceptual models were developed and presented in ERAs by Solomon *et al.* (1996); Cormier *et al.* (2000); USEPA (2002a; b); and Salihoglu and Karaer (2004). The conceptual models linked stressor exposure with the environmental processes. The conceptual model

developed by USEPA (2002b) (Figure 7) in their ERA presented a conceptual flow diagram that linked sources, stressors and effects, with endpoints. Often stressor sources were related to land uses in the defined area and contamination described by environmental fate principles. The conceptual models in each case played a vital role in tracking chemical stressors and highlighting potential environmental sources of concern to be investigated.

Overall, conceptual models generate a series of hypotheses that describe the expected relationships and interactions that exist between the ecosystems at risk, identified potential stressors, and ecological effects and set the scope for the analysis phase. It also provides decision makers with a record of options of the local and scientific experts and references upon which the opinion is based.



**Figure 7.** Conceptual model of the Waquoit Bay watershed ecological risk assessment presented by USEPA (2002b). Bold items are human activities, rectangles represent sources of stressors, hexagons are the specific stressors to the system, trapezoids represent the effects of those stressors, and the ellipses indicate specific endpoints that are affected.

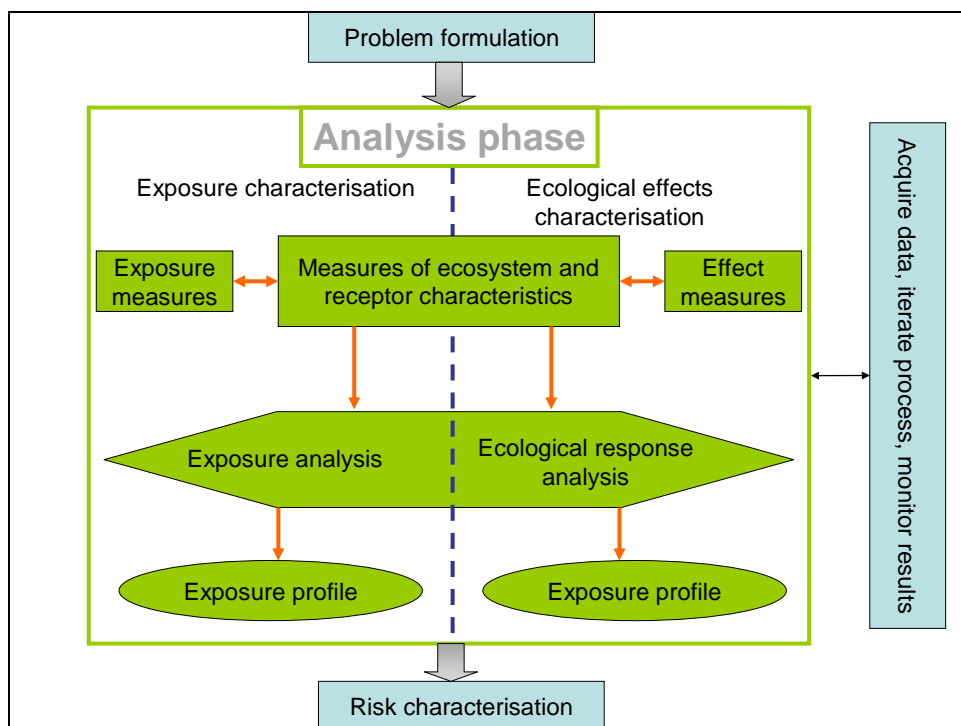
### 5.2.6 Outcome of problem formulation

In summary, problem formulation phase provides assessors with a characterisation of the defined environment and stressors to be investigated; a series of questions to be assessed; assessment endpoints that identify the properties of the valued ecological resources; identified measurements needed to quantify risks or impacts; a conceptual model describing relationships with stressors, ecological resources, and effects; and an analysis plan to support the proceeding phase of the ERA.

### 5.3 Analysis phase

The analysis phase (Figure 8) utilises the relationships presented in the conceptual model and develops them further, focusing on the most important stressors, their exposure pathways and ecological effects (Serveiss *et al.* 2000; USEPA 1998). This is achieved by taking targeted measurements, modelling or extrapolation from field or laboratory data to describe existing conditions and subsequently characterise the exposure and effects. In essence, the phase provides a greater insight to the relationships that exist between the stressors behaviour in the environment and the effects on organisms that result from exposure (Serveiss *et al.* 2000; USEPA 1998; 2002a; b). As the analysis proceeds, interim findings are often presented to environmental managers and community groups to follow-up on the appropriateness of the output that will ultimately formulate management decisions.

In taking measurements of existing conditions, a quantitative approach is often not possible for each exposure pathway or effects scenario. The quantification of risks may involve targeting single species or chemicals that represent simplified scenarios as opposed to complex and variable situations observed in the environment. It is therefore important to acknowledge that ecosystems are exposed to multiple stressors and are also likely to exhibit incomplete information. Subsequently, scientific judgement and a weight of scientific evidence approach is often adopted to address information gaps in estimating exposure or effects (Serveiss *et al.* 2000; Solomon *et al.* 1996; USEPA 1998; 2002a; b). Further, the risk analysis may focus on the ecological effects posed by a stressor of concern, or seek associations with multiple stressors. As a final step, the uncertainty of all the information utilised must be conveyed.



**Figure 8.** Conceptual diagram of the analysis phase of ERA (taken from Serveiss *et al.* 2000).

### 5.3.1 Characterising exposure

Exposure is assessed by associating agrochemicals with sources, describing their temporal and spatial distribution in the catchment, and the extent to which stressors are in contact with the organisms (Serveiss *et al.* 2000; Solomon *et al.* 1996; USEPA 2002a; b). Exposure of a stressor is estimated by obtaining data through direct measurement, or fate and transport modelling (Serveiss *et al.* 2000; USEPA 1998). The magnitude and spatial and temporal distribution of a stressor is also considered when characterising the relevant exposure pathways as well as in the development of a quantitative exposure profile.

A combination of monitoring and modelling is often used to characterise temporal and spatial distribution in the environment, and the extent to which stressors are in contact with the organisms. In situations where the aim is to characterise impacts on an ecosystem following a disturbance and to evaluate ecological populations prior to the event is often necessary. A number of considerations are made when characterising a stressor, including:

- What are the sources of the stressor?
- What is the spatial and temporal distribution of the stressor?
- Are there additive stressors (secondary stressors) associated with the original stressor?
- How does the timing and location of the stressor interact with the ecological resource?

A comprehensive exposure profile is produced while characterising ecological exposure. The aim of an exposure profile is to quantify the spatial and temporal patterns of the stressor occurrence and highlight the adverse effects experienced by ecological resources. Exposure may be characterised through direct field measurement of the stressor or through modelling (Holvoet *et al.* 2007; Serveiss *et al.* 2000; USEPA 1998). Initially, exposure is characterised using a conceptual model to highlight important environmental fate and transport processes. Often assessing and conceptualising environmental fate and transport of chemicals is necessary to support investigations. The potential for a chemical to be transported in a defined media is often described using fugacity principles. Fugacity is a concept that describes the escaping tendency of a chemical from a defined phase, solid, liquid and gas. Models have been developed to support these principles, and are commonly known as fugacity or multimedia models. The environmental complexity these models can accommodate is dependent on available information for the defined environment.

### 5.3.2 Sampling approaches

Characterising exposure in ecosystems using direct quantitative methods is commonplace. A number of different methods are used in the quantification process, such as grab or passive sampling. In both cases, spatial and temporal considerations are made in designing a sampling regime, such that adequate characterisation of the catchment can be made.

Grab sampling is a method often carried out to characterise stressor exposure in a catchment. These samples are widely acknowledged to effectively characterise concentrations for one point in time through direct analysis. The method generally involves inserting a container under or down-current of a discharge with the container opening facing upstream below the water surface. A number of samples are taken in sequence to characterise concentrations temporally, and where possible at a number of different points to account for spatial variation. For example, El-Kabbany *et al.* (2000) characterised exposure for a range of organic pesticides in the El-Haram Giza region of Egypt, by carrying out targeted sampling in various reaches of the Giza region. Comoretto *et al.* (2007) investigated pesticide exposure in the Rhône River delta, France and was able to report on seasonal and spatial variation in exposure. Although the grab sampling may adequately characterise the exposure being exhibited in the environment temporally and spatially, the time and resources required to execute an adequate investigation is exorbitant and costly.

Another sampling approach involves the use of passive samplers. Passive samplers are increasingly being used in the determination of time integrated pesticide concentrations



over extended periods of time. They range in design and characteristically exhibit material that is able to sorb and concentrate a range of analytes in question (Alvarez *et al.* 2004; Hyne *et al.* 2004; Petty *et al.* 2004). The chemical nature of the sorbent varies according to the chemical nature of the target contaminant, which the target analyte generally has a high affinity. The devices are typically immersed in the water over an extended period of time, and it is anticipated that a range of target analytes will sorb to the sorbent (Alvarez *et al.* 2004; Hyne *et al.* 2004; Petty *et al.* 2004). When the sampler is removed, the analytes are eluted from the sorbing material and analysed (Alvarez *et al.* 2004; 2000; Hyne *et al.* 2004; Petty *et al.* 2004). The residue that is eluted is determined as a load and is considered to be an integral of the concentration during the time the sampler is spent immersed in the water body. The time interval average concentration is estimated from the load sorbed to the sorbent and the average discharge of the reach (Alvarez *et al.* 2004; Hyne *et al.* 2004; Petty *et al.* 2004). Hyne *et al.* (2004) used a passive sampler to estimate the concentration of  $\alpha$ -endosulfan,  $\beta$ -endosulfan, endosulfan sulfate and chlorpyrifos-ethyl for periods of 7-22 days in the Gwydir River and Namoi River catchments. Petty *et al.* (2004) investigated, using passive samplers, time integrated concentrations of organochlorine pesticides, polycyclic aromatic hydrocarbons, organophosphate pesticides, and pharmaceutical chemicals in waste water discharging into a constructed wetland located in the Columbia, Missouri, USA. Hyne *et al.* (2004) and Petty *et al.* (2004) found that the passive samplers used were able to predict within reasonable limits average pesticide concentrations despite the wide variation in river-water concentrations through time. Hyne *et al.* (2004) further concluded that the devices cannot be used for direct measurement of snapshot concentrations of pesticides entering rivers and waterways. However, the major benefits of using passive samplers has generally been found to minimise the cost and maximise the precision and breadth of application and was suitable for systems with temporally variable concentrations of contaminants of concern (Alvarez *et al.* 2004; Hyne *et al.* 2004; Petty *et al.* 2004).

Overall, two common methods, grab and passive sampling, are used to characterise exposure in river reaches. However, it appears suitability of the methods is directed at the required detail in characterising exposure. Further to this, the time and cost are factors considered in method selection.

#### 5.3.4 Modelling approaches

It is often undesirable to carry out extensive field monitoring to characterise exposure due to the large financial and human resource requirements. Where sufficient monitoring data may

be lacking to support the ERA, environmental fate and transport models may be used to predict stressor concentrations in the catchment (Holvoet *et al.* 2007). A number of different fate and transport models have been developed to predict the exposure of agrochemicals in surface waters at the catchment-scale, which some are presented in Table 2. The scope and complexity of these models vary depending on their intended purpose (Coulibaly *et al.* 2004). The success of a model at predicting exposure is ultimately dependent on the reliability of input data, including characterisation of contaminant sources and landscape parameters (Coulibaly *et al.* 2004).

**Table 2.** Some useful catchment-scale models.

Model	Reference
PRZM (Pesticide Root Zone Model)	
GLEAMS (Groundwater Leaching BasinBox)	(Hollander <i>et al.</i> 2006)
SWAT (Soil and Water Assessment Tool)	(Neitsch <i>et al.</i> 2002)
HSPF	
TREX	(Velleux <i>et al.</i> 2008)
EXAMS	
ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation)	(Beasley <i>et al.</i> 1980)

Exposure models have been used by a number of authors to supplement exposure data shortfalls in ERA. Solomon *et al.* (1996) predicted and compared exposure of atrazine in runoff water from treated fields using the USEPA's PRZM (Pesticide Root Zone Model) and the USDA's GLEAMS model. Parker *et al.* (2007) simulated atrazine, metolachlor, and trifluralin transport in the Sugar Creek sub-catchment of the White River catchment, Indiana, United States using three catchment-scale models, the Soil Water Assessment Tool (SWAT), the Nonpoint Source Model (NPSM), a modified version of the Hydrologic Simulation Program-Fortran (HSPF), and the Pesticide Root Zone Model-Riverine Water Quality (PRZM-RIVWQ). USEPA (2002b) used a nitrogen loading model (NLM) to estimate the amount of nitrogen delivered to an estuary in Waquoit Bay, Massachusetts, USA. Multimedia models have also been developed that predict the fate of chemicals based on fugacity principles (see Mackay 2001).

More recent developments in fate modelling account for spatial variability. For example, Coulibaly *et al.* (2004) presented a GIS-based multimedia catchment model using available GIS data, chemical release information and local monitoring networks to assess the fate of trichloroethene (TCE) within the Passaic River Watershed, USA. A spatially distributed model to assess watershed contaminant transport and fate; called the Two-dimensional, Runoff, Erosion, and Export model (TREX); was developed by Velleux *et al.* (2008). Exposure may be further characterised based on chemical use in catchments. Solomon *et al.* (1996), and USEPA (2002a; b) characterised the spatial and temporal distributions of their stressor sources based on lines of evidence. Furthermore, they characterised land practices for their influence on exposure trends. Data requirements to support fate and transport modelling are very intensive. Table 3 summarises useful databases to support GIS modelling in Australia.

**Table 3.** Useful information sources for GIS exposure modelling in Australia.

Information	Location
Digital elevation model (DEM)	United States Geologic Survey: Seamless Server ( <a href="http://seamless.usgs.gov/index.php">http://seamless.usgs.gov/index.php</a> )
Land use	Australian Natural Resources Data Library ( <a href="http://adl.brs.gov.au/anrdl/php/basic_search.php">http://adl.brs.gov.au/anrdl/php/basic_search.php</a> )
Climate data	Bureau of meteorology
Soil information	Geoscience Australia; CMAs or State Natural Resource organisations.
Chemical information	APVMA, Farmers and natural resource managers

Although there are a range of models that have the capacity to support catchment-scale ERA, monitoring data is required to validate and calibrate the model outputs. Parker *et al.* (2007) revealed the necessity of formulating monitoring programs to support exposure modelling requirements. Such planning allows for the collection and recording of additional data requirements and may impact on the sampling schedule (Parker *et al.* 2007). Subsequently, uncertainty in model estimates needs to be estimated to minimise the bias in an ERA.

In conclusion, exposure can be characterised by monitoring ecological resources or carrying out fate and transport modelling. A range of fate and transport models are available

to characterise exposure, however such models need to be adapted to the environmental conditions which they are to emulate. Furthermore, considerations for uncertainty need to be made when conducting modelling.

### 5. 3.5 Characterising effects

Ecological effects are determined by stressor-response relationships, which include evaluations for evidence of causality, and linking the effects to the assessment endpoints prescribed in problem formulation. Devising effects relies on professional judgement relative to other analysis steps, as the selection of data is highly subjective with regard to selecting data sources or decisions on filling information gaps (Serveiss *et al.* 2000; Solomon *et al.* 1996). The relevance that available data holds is manifested in the indicators chosen in the problem formulation, as well as its integrity. Statistical methods or mathematical models may be used to quantify and summarise a relationship of the stressor to an ecological resource. Extrapolation from between taxa, laboratory to field, and field to a field in a different region or scale, is often necessary. Exposure toxicity to various organisms culminates where high enough concentration can induce some form of effect toward certain organisms. A number of methods have been developed to investigate the toxic effects that various chemicals pose. The methods range in their complexity with regards to emulating likely exposure conditions observed in the environment. Simple laboratory exposure experimental types include static, replacement and flowthrough. More complex ecosystem methods, known as cosms, account for effects posed on whole ecological communities resulting from exposure.

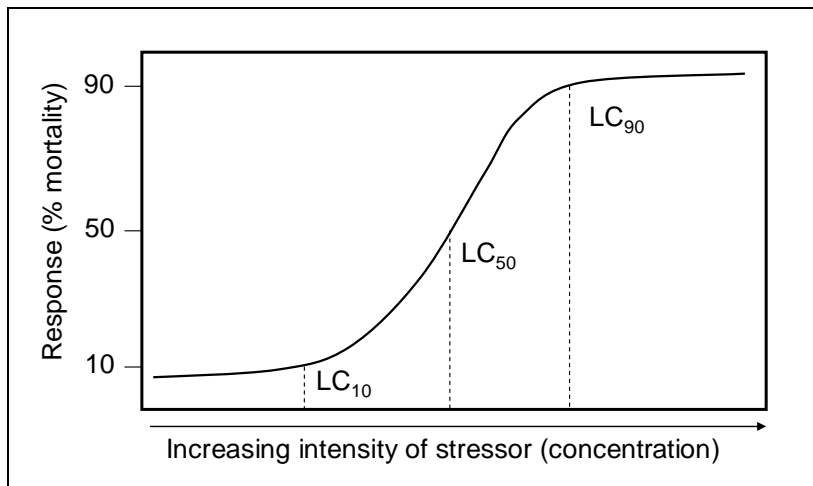
Effects data used in ERA is often synthesised from literature and databases (e.g. USEPA's ECOTOX Database). Reference to assessment endpoints is made when selecting appropriate effects data. Considerations for cascade effects on the ecosystem, where the effect imposed on one organism may lead to detrimental effects on a range of organisms in the ecosystem (e.g. algae and fish), is also made where necessary. Solomon *et al.* (1996), and USEPA (USEPA 2002a; b) accounted for a range of effects and further related these to cascade of stressor effects posed to the various trophic levels of the aquatic ecosystem investigated. For example, it may be that the more sensitive species may in fact lead to the demise of more resilient species, e.g. contamination of an ecosystem by the herbicide diuron may kill plant species, which more resilient species of fish rely on for food and oxygen. Subsequently experimental procedures used in effects experiment need to be evaluated for their ability to emulate environmental scenarios when selecting data. This often includes comparing static, replacement, and flowthrough procedures. Other effects tests account for

more complex scenarios, such as the interactions that occur within ecosystems, the occurrence of multiple-stressors, and time-variable exposures. Examples of these include microcosms (e.g. Hanson and Solomon 2002) or mesocosms (e.g. Downing *et al.* 2008; Nelieu *et al.* 2005; Pennington *et al.* 2004) and pulse studies (e.g. Downing *et al.* 2008; Vallotton *et al.* 2008). In particular, the use of microcosms has allowed the measurement of the environmental effects measures, such as NOAEC<sub>community</sub> (No Observed Adverse Effect Concentration for a community of organisms), in ecological communities (Campbell *et al.* 1999; Hill *et al.* 1994; Van den Brink *et al.* 1996). Overall, there are a number of ways to account for effects, however choosing effects data that closely reflects conditions observed in the environment appears critical.

### 5.3.6 Stressor-response relationships

Characterising the stressor-response relationships is where quantitative analysis of the exposure and effects takes place. A number of statistical methods have been developed to support this component of the analysis phase.

The stressor-response profile relates the extent of the effect to the magnitude, duration, frequency, and timing of the exposure. For assessments that are limited to a single stressor and effect, the relationship is often expressed as a stressor-response curve (Figure 5), and summarised as a single reference value such as an LC<sub>50</sub> (lethal concentration effecting 50% of the test population) or EC<sub>10</sub> (concentration causing an effect to 10% of the population) depending on the scenario being described and the best approach for its presentation. A stressor response curve is a graph depicting a quantitative representation of the relationship between a stressor (such as a pesticide concentration in the water column) and an ecological effect (such as mortality of a given fish species if exposed to different concentrations of the pesticide). The result is a distribution displaying percentage of the test population affected resulting from increasing exposure concentration (Figure 5). Standard testing procedures have been developed to support the chronic and acute response of fish (OECD 1984; 1992), aquatic plants (OECD 1998b; 2006), vertebrates and invertebrates (OECD 1998a; 2004) resulting from exposure.



**Figure 5.** Simplified stressor response relationship of response (% mortality) and increasing intensity of stressor (concentration) (optimised after Serveiss *et al.* 2000).

The response analysis in ERA relates the strength of the association exhibited between the stressors and the assessment endpoints and indicators. At best, a cause-effect relationship is observed, however watershed complexity and limited available data often constrain this outcome (Serveiss *et al.* 2000). In ideal situations, the stressor-response relationship has the capacity to relate the magnitude, duration, frequency and timing of the exposure in the catchment to the effects; however documenting general relationships of sources or stressors and their effects can be achieved for multiple stressor assessments of watersheds.

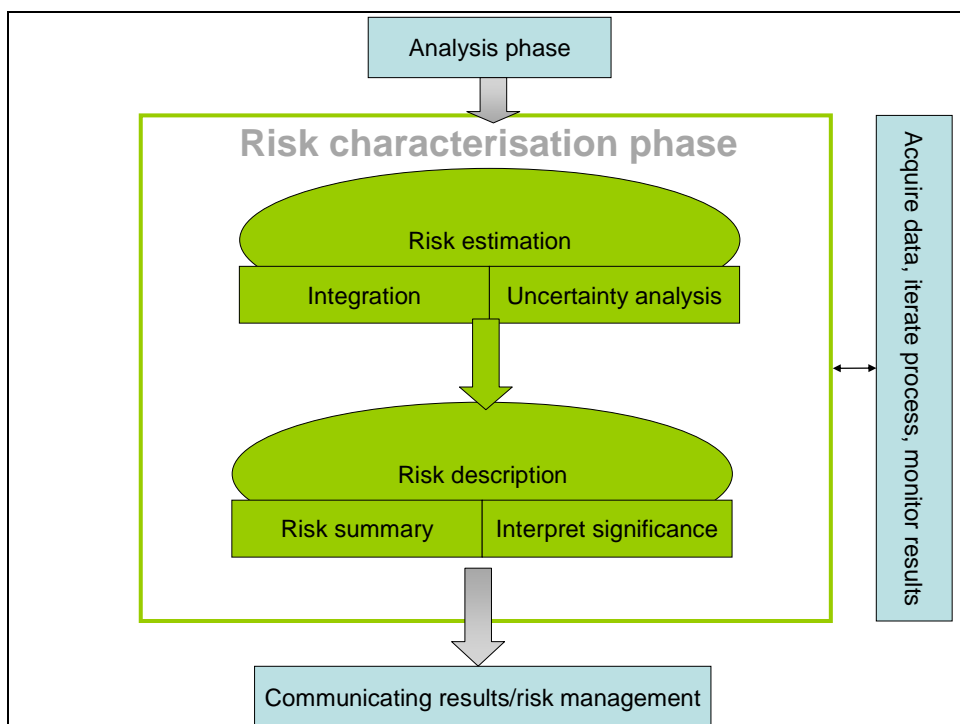
When characterising effects posed to multiple species exposed to the same stressor, a common method is to construct a species sensitivity distribution (SSD). An SSD is a statistical distribution that combines all single species effects data and displays it as a cumulative distribution function (Posthuma *et al.* 2002). Often there may be differences exhibited between groups of organisms as a result of the stressor mode of action, e.g. plants and vertebrates. Theory on SSDs is described by Posthuma *et al.* (2002) and Maltby *et al.* (2005). The hazardous concentration that is acceptable in Australia is 5% of species (ANZECC 2000). That is, it is acceptable that 5% of species be affected before risk management is warranted.

Overall, the analysis phase produces a characterisation of exposure. The phase provides a description of the patterns of stressor occurrence in the catchment; ecological response analyses that describes the effects posed on the ecological resources identified in the

problem formulation phase; quantitative stressor-response relationships; and documented assumptions and uncertainty used in the analyses.

#### 5.4 Risk characterisation phase

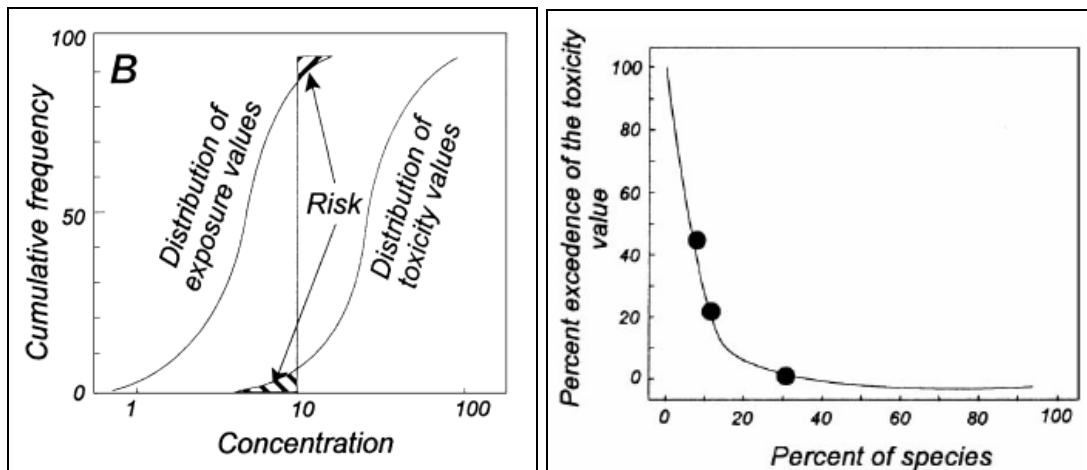
Risk characterisation is the final phase in ERA. In risk characterisation, the likelihood and significance of adverse effects resulting from exposure of stressors determined. Two major steps are taken in this phase: risk estimation and risk description (Figure 6). Finally, a risk assessment report is prepared for managers to support decisions that are scientifically sound and are based on defensible assessment conclusions.



**Figure 6.** Conceptual diagram of the risk characterisation phase (reproduced from USEPA 1998)

Risk estimation involves integrating the exposure profiles and stressor-response profiles developed in the analysis phase. Uncertainties that arise from the estimation of risk are also characterised. A range of approaches may be used such as hazard quotient (e.g. Nabholz 1991); comparing statistical distributions of exposure and effect (e.g. Solomon *et al.* 1996); or conducting modelling (e.g. Burmaster and Anderson 1994; Campbell *et al.* 2000; Park *et al.* 2008; Pollino *et al.* 2007). Nabholz (1991); Solomon and Sibley (2002); and Tannenbaum *et al.* (2003) revealed that the most common approach in ERA is the hazard

quotient method. This is simply a ratio of the highest measured or predicted exposure concentration with the most sensitive test species exposure toxicity concentration. Where the quotient exceeds one indicates “risk”. However, Tannenbaum *et al.* (2003) argued that this simple relationship is not a measure of risk as it lacks the probabilistic paradigm. Solomon and Sibley (2002) further argued that single species tests do not consider the presence of multiple species, each with a particular sensitivity or the interactions that can occur between these species in a functioning community, and is therefore unsuited in ERA. The hazard quotient is increasingly being used to gauge the necessity for further exposure assessment where the quotient exceeds one. Alternatively, Solomon *et al.* (1996); and Hanson and Solomon (2002) constructed species sensitivity distributions to characterise the effects posed to a range of organisms by atrazine, tributyltin and monochloroacetic acid, respectively. By relating exposure distributions to SSD’s they were able to ascertain which species were likely to be affected, and further assign a probability of exceeding a threshold via a joint probability curve (Figure 7; Solomon *et al.* 2000). Other methods of characterising risk include Bayesian Networks (Pollino *et al.* 2007), Monte Carlo (Burmester and Anderson 1994; Campbell *et al.* 2000; Hope 2001) and other complex statistical analyses.



**Figure 7.** Relating exposure and toxicity (also known as an SSD) distributions to characterise risk (left); and a joint probability curve that highlights the level of risk (right) (taken from Solomon *et al.* 2000).

Estimation of risk also entails uncertainty. Uncertainties that have arisen throughout the ERA must be characterised. The analysis may include measurement data (inappropriate, imprecise or too few measurements), conditions of observation (such as extrapolating from laboratory tests to field predictions), or limitations of models (e.g., oversimplifying complex



ecological processes). In the case of limited data, qualitative assessments may be used to rank risks using ranking criteria based on professional judgement and assigning categories, such as low, medium and high. Common statistical means of quantifying uncertainty in ERA often utilises Monte Carlo analysis (Burmaster and Anderson 1994; Hope 2001). Adequate characterisation of the uncertainty enhances the credibility of the conclusions drawn and provides certainty in decision making.

Risk description is where conclusions about the characterisation phase are made through a summary of ecological risk and elucidation of ecological significance. Summarising risk involves assigning definitions to risk in the form of a quantitative statement (e.g. there is a 20% chance that 20% of fish will die in the catchment due to runoff polluted by a defined agrochemical). Including weight of evidence discussion to support the conclusion that may cover the quality of the data, supporting information, and evidence of impact, is also crucial in supporting the risk description. If strong agreement among multiple lines of evidence can be made increases the confidence in the conclusions, however differences in conclusions warrants further discussion. Additional analyses may also be identified to improve the assessments certainty.

Overall, risk characterisation is intended to provide a clear description of the calculated risks. The level of risk is ultimately prescribed using statistically sound methods that also provide insight into the level of uncertainty associated with the characterised risk. The characterisation also draws conclusions using all the information gathered and highlights any uncertainty faced during the course of ERA.

### *5.5 Risk communication and management*

The information conceived throughout the ERA should be collated and presented in a report. Reporting should be in the form of a document that is freely available, as well as providing summary reports for community groups highlighting important findings and giving a final presentation of findings to the catchment managers and stakeholders. Final discussion meetings should be held with the stakeholders highlighting important findings and allow for questions.

Risk management involves integrating science-based assessment with economic, social, legal, and political factors affecting management decisions and actions in the catchment. The management of risk should be site-specific, following the utilisation of exposure models that predict the fate and transport of chemicals that account for spatial and temporal exposure dynamics.

## **6. THE SCOPE FOR CATCHMENT-SCALE ERA IN AUSTRALIA**

Immense complexity is entrenched in the ERA process alone. Current research is focussed around accounting for as much environmental complexity as possible. Current developments in ERA are being seen through innovations in modelling and different ways of displaying risk. The current focus is to enhance the complexity of estimating and displaying exposure such that the reliability of ERA is enhanced. Accounting for spatial and temporal variability, a site specific approach applied to GIS for agrochemical management seems a logical approach.

Ecological risk assessment methods are designed to assimilate scientific information into environmental decisions. Catchments are mostly isolated hydrologic ecosystems and are therefore a logical unit for environmental management. Management of water resources in Australia occurs at the catchment-scale; however management of agrochemicals occurs at the field- or farm-scale. A practical means for applying agrochemical ecological risk assessment to the catchment-scale in Australia is currently not developed. A catchment-scale approach to ecological risk assessment would allow for environmental managers to direct necessary resources to manage agrochemicals that are affecting groundwater and surface water resources in an effort to maintain the health of ecosystems and communities.

The off-site movement of agrochemicals in run-off, from both cotton and sugarcane production, has not been accurately assessed at the catchment-scale. Quantifying the movement of water off-farms and subsequent contamination of surface waters would ultimately impact on end-of-catchment contamination risk to ecological resources. Recently, the risk of contamination of the Great Barrier Reef Marine Park, by a range of agrochemicals, has led to the development of Marine Water Quality Guidelines by the Great Barrier Reef Marine Park Authority. Identifying the contamination risk of a range of environmentally significant sites, such as the Great Barrier Reef Marine Park, will indicate specific needs for management, whether it is at the catchment- or farm-scales.

Registration of agrochemicals in Australia applies certain conditions to a chemical's application in agricultural production. However, these conditions do not discriminate spatial and temporal variability of catchments across Australia, conditions that are likely to influence risk. It is therefore important to distinguish catchments on the basis that they exhibit differing climatic and physical characteristics as well as spatial and temporal variability. Such attributes are likely to have a significant impact on the factors that influence the fate of agrochemicals, such as runoff, drift, sorption, etc. and impact on their exposure and ultimately risk to ecological resources. It is therefore necessary to account for these

differences in risk characteristics, hence the necessity in making catchment-scale ecological risk assessment site-specific by utilising a range of spatial methods to characterise exposure. The identification of risk “hotspots” in a catchment will allow for catchment managers and farmers to manage risk on the farms relative to nation-wide regulations.

Following characterisation of risk using various statistical methods highlighted earlier, displaying risk spatially is very useful. Where sufficient data exists and calculated risk has been georeferenced, risk may be interpreted at larger spatial scales by being displayed on maps produced using GIS. This method of displaying spatial risk is likely to significantly aid in risk management and is currently one of the major areas under development in ERA.

It would appear that no consistent, rigorous ERA framework that environmental managers can follow is available in Australia. Such scientifically rigorous and defensible procedure would eliminate the current political issues surrounding ecological impact issues and provide a heightened level of certainty in the management of agrochemicals. Furthermore, as the natural resources are increasingly being managed at the catchment-scale in Australia, an ERA process that reflects this scale would be logical. The proficiency of such a tool may be further enhanced by incorporating ERA with GIS to identify hotspot risk areas.

## **7. CONCLUSION**

This review has highlighted the important role that agrochemicals play in the agricultural landscape, with special reference to the Australian cotton industry. Agrochemicals were shown to deliver significant benefits to production, a prospect that has enhanced profitability. Although their intended application is to protect crops and enhance productivity, a number of environmental mechanisms operate to degrade pesticides and also transport chemicals from the site of application. Subsequently, agrochemicals were found to pose environmental hazard to non-target organisms in catchments. The common means that agrochemicals are managed in catchments around the world has involved the use of ecological risk assessment framework adapted to the catchment-scale, and it was further found that such a framework is lacking in Australia.

As the management of natural resources in Australia has been described to operate at the catchment-scale, the management of agrochemicals was found to not operate at this scale. Subsequently, a catchment-scale ecological risk assessment framework developed by the USEPA was reviewed. The framework was found to consist of three key phases planning, analysis and risk characterisation. The different components were assessed for different

contributions to the ERA process, drawing on examples of ecological risk assessments published in the literature. A current development in ERA involves developing ways of determining risk at the catchment scale, utilising spatial models that have the ability to predict chemical movement at the catchment-scale. Further developments have involved applying this information to a GIS platform so that maps highlighting hotspot areas can be produced to support management decisions. It was concluded that such a framework, involving the utilisation of GIS, would be a scientifically defensible and reliable tool in the management of agrochemicals at the catchment-scale in Australia.

Overall, it can be concluded that agrochemicals used in agricultural practices in Australia is likely to pose a threat to catchment ecological communities. One of the ways to support the management of agrochemicals is through the development of a scientifically rigorous catchment-scale ecological risk assessment framework. Further, the current developments in catchment-scale ERA involves the use of geographical information systems to highlight areas in catchment that display high and low risk. Such a method would provide natural resource managers with a powerful tool in the management of agrochemicals in Australia.

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