WETLAND ECOLOGY AND CONSERVATION

Risk Identification and Assessment

Ecological risk assessment of the herbicide tebuthiuron in northern Australian wetlands[1]

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1 Introduction

Background

The herbicide tebuthiuron has been used widely in the Northern Territory of Australia for control of the wetland weed, *Mimosa pigra* (Mimosa), since the late 1980s. Mimosa is an opportunistic and aggressive weed, forming dense mono-specific stands in tropical wetland habitats and replacing native vegetation (Lonsdale et al 1995). Thus, there is a need to effectively control and manage Mimosa in northern Australian wetlands. However, the control measures themselves may well impart some adverse impact on the local environment. Ideally, potential adverse impacts of control measures should be assessed prior to their implementation. Where this has not occurred, appropriate assessments should be carried out as a priority. While the long-term goal for the effective management of Mimosa in northern Australia is the establishment of a successful biological control program (Forno 1992), it is acknowledged that this will need to be used in conjunction with chemical and mechanical methods (Environment Australia 1997). Therefore, the current use of herbicides will continue in the long-term, and it is imperative that their risks to the local aquatic environment are assessed and understood. Historically, tebuthiuron has been the most commonly used herbicide for Mimosa control in northern Australia, and for this reason was the focus of this assessment.

Aims and working hypotheses

The study aimed to provide a quantitative estimate of the ecological risks of tebuthiuron to the freshwater fauna and flora of northern Australian wetlands.

The following two working hypotheses were assessed:

1. That tebuthiuron may result in direct adverse effects to native freshwater biota at the site and downstream of treated *M. pigra* infestations, potentially resulting in adverse effects to community structure and function; and
2. That long-term and/or delayed effects to native freshwater biota may occur as a result of the residual properties of tebuthiuron.

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[1] More detailed discussion of this research is provided in: Camilleri C, Markich S, van Dam R & Pfeifle V 1998. *Toxicity of the herbicide Tebuthiuron to Australian tropical freshwater organisms: Towards an ecological risk assessment*. Supervising Scientist Report 131, Supervising Scientist, Canberra. & van Dam RA, Camilleri C & Markich SJ 1999. Ecological risk assessment of the herbicide Tebuthiuron in northern Australian wetlands. *Proceedings of the EnviroTox'99 International Conference*, Geelong, Australia, 7–10 February 1999.

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Approach

The ecological risk assessment generally followed the probablistic approach recommended by the U.S. Environmental Protection Agency (1998). Following the problem formulation stage (partially addressed above, but elaborated upon in van Dam et al [2001]), the assessment involved the following three major steps: *effects characterisation*, *exposure characterisation* and *risk characterisation*. A final section identifies some *management implications*.

2 Effects characterisation

Effects characterisation involved assessment of the acute or chronic toxicity of tebuthiuron to five local freshwater species (three animals and two plants), and comparison of the results with toxicity data derived for northern hemisphere species. Table 1 summarises the results of the toxicity tests. Freshwater plant species were about 2 to 3 orders of magnitude more sensitive to tebuthiuron than the animal species. *Lemna aequinoctialis* was the most sensitive species tested, while *Mogurnda mogurnda* was the least sensitive, although the latter estimate was based on an acute response.

Table 1 Summary of tebuthiuron toxicity to five tropical freshwater species

Test organism	Test duration (acute/chronic; endpoint)	EC_{50} (mg L^{-1})	NOEC (mg L^{-1})	LOEC (mg L^{-1})
Chlorella sp. (green alga)	72 h (chronic; cell division rate)	0.25	0.092	0.19
Lemna aequinoctialis (duckweed)	96 h (chronic; plant growth)	0.14	0.05	0.1
Moinodaphnia macleayi (water flea)	3 brood (chronic; reproduction)	134	20	40
Hydra viridissima (green hydra)	96 h (chronic; population growth)	150	50	75
Mogurnda mogurnda (purple-spotted gudgeon)	96 h (acute; survival)	214[a]	200	225

[a] LC_{50}

In general, there were no major differences in the acute and chronic toxicity of tebuthiuron between northern hemisphere and Australian tropical aquatic species. The acute LC_{50} values of tebuthiuron for northern hemisphere temperate freshwater fish (112 – >160 mg L^{-1}) tended to be slightly lower than the Australian tropical freshwater fish, *M. mogurnda* (Caux et al 1997), although the maximum difference was less than two-fold. Similarly, chronic toxicity values for algae varied a little between the data sets, but were less than an order of magnitude different. A comparison could not be made for hydra, as no comparable temperate data were available.

Based on the available literature, it appears that the toxicity of tebuthiuron to a limited number of Australian tropical freshwater organisms is similar to that of northern hemisphere temperate species. Given this, it was considered appropriate to incorporate the existing, northern hemisphere toxicity data with the local species toxicity data for the risk characterisation component of the risk assessment.

3 Exposure characterisation

Exposure characterisation involved the use of historical field monitoring data of tebuthiuron concentrations following applications of tebuthiuron to a large Mimosa infestation on the Oenpelli floodplain, western Arnhem Land in 1989 (1500 kg tebuthiuron to 1000 ha Mimosa,

Parry & Duff 1990) and 1991 (12 000 kg tebuthiuron to 5800 ha Mimosa, Cook 1992). Tebuthiuron concentrations in surface water ranged from 0.002 to 2.05 mg L^{-1}. The highest concentration of 2.05 mg L^{-1} was detected three days after application. Tebuthiuron was still measurable in surface water three, four and five months following application, with the highest concentrations at these time points being 0.168, 0.037 and 0.034 mg L^{-1}, respectively.

4 Risk characterisation

Risk characterisation involved the comparison of cumulative lognormal probability distributions of environmental tebuthiuron concentrations and species sensitivity to tebuthiuron. The degree of overlap between distributions of species sensitivity and environmental concentrations is used to estimate the risks to aquatic biota. Risks were estimated for freshwater plant chronic toxicity (fig 1A), invertebrate and vertebrate chronic toxicity (fig 1B), and vertebrate acute toxicity (fig 1B). The probability of the environmental concentration of tebuthiuron exceeding the 1st, 5th and 10th centiles of the species sensitivity distributions, are shown in table 2. These values correspond to the probability of 1, 5 or 10% of species being affected.

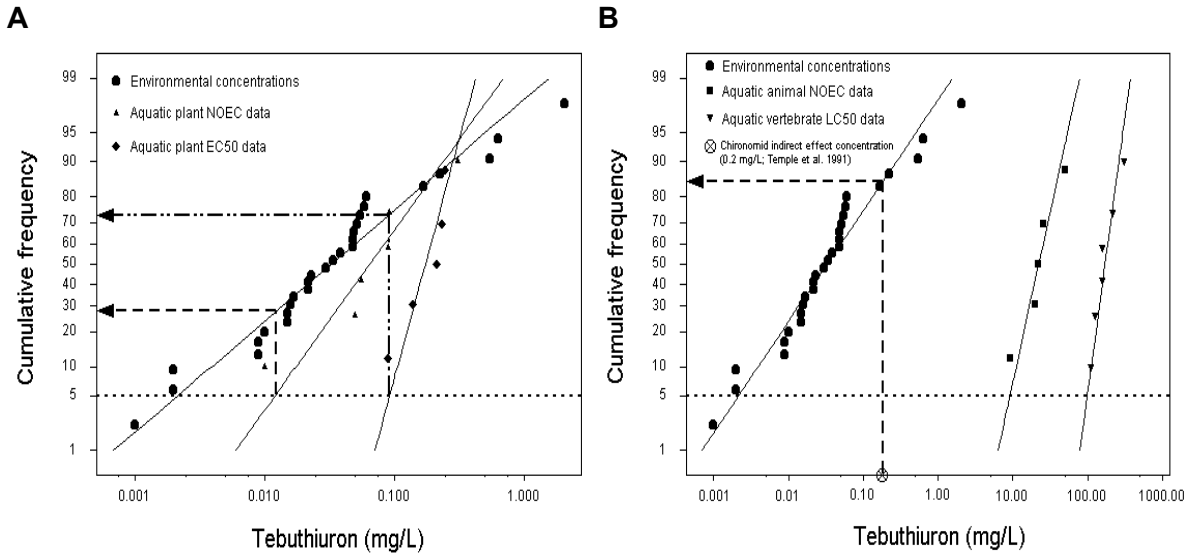

Figure 1 Comparison between the distribution of environmental tebuthiuron concentrations and (A) chronic plant sensitivity distributions for tebuthiuron based on NOEC data and EC_{50} data, and (B) chronic animal sensitivity and acute vertebrate sensitivity distributions for tebuthiuron based on NOEC and LC_{50} data, respectively. Broken line arrows in (A) indicate the point of overlap at the 5th percentile of the species sensitivity distributions with the distribution of environmental tebuthiuron concentrations. The broken line arrow in (B) indicates the point of overlap of a reported indirect effect on chironomids (Temple et al 1991) with the distribution of environmental tebuthiuron concentrations.

Table 2 Risks of tebuthiuron to freshwater species in northern Australian wetlands

Scenario	Probability of x% of species being affected		
	10%	**5%**	**1%**
Plant chronic effects			
NOEC data	65% (0.018; 0.006-0.05)[a]	73% (0.012; 0.003-0.04)	85% (0.006; 0.001-0.03)
EC_{50} data	24% (0.106; 0.067-0.167)	27% (0.092; 0.055-0.155)	32% (0.071; 0.037-0.136)
Animal chronic effects	<1% (11.1; 5.9-20.6)	<1% (9.1; 4.4-18.8)	<1% (6.3; 2.5-15.7)
Vertebrate acute effects	<1% (111; 78-159)	<1% (98; 65-148)	<1% (79; 47-131)

[a] Values in parentheses represent the corresponding tebuthiuron concentration (mg L^{-1}) and its associated 95% confidence limits.

Freshwater plant chronic effects

As expected, risks of tebuthiuron to freshwater plants were far greater than to animal species. Based on the tebuthiuron levels measured in water on the Oenpelli floodplain following application in 1989 and 1991, the probability of freshwater plant species experiencing chronic effects can be considered high (table 2, fig 1A). To demonstrate the relevance of the persistence of tebuthiuron in surface water, the comparison of effects and exposure distributions was repeated for freshwater plants using only tebuthiuron concentrations measured three months or more following application (fig 2). The risks of tebuthiuron to freshwater plant species remained high for some time following application, with the probability of at least 5% of species experiencing chronic effects still approximately 63% (based on NOEC data).

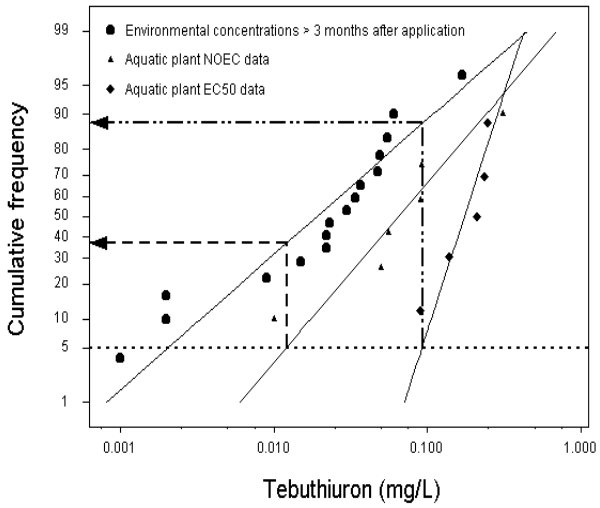

Figure 2 Comparison between the distribution of environmental tebuthiuron concentrations measured ≥3 months following application and the chronic plant sensitivity distributions for tebuthiuron based on NOEC data and EC_{50} data

Freshwater animal chronic effects and vertebrate acute effects

The risk of chronic direct effects to freshwater animal species (invertebrates and vertebrates) can be considered low, with the concentrations estimated to affect even 1% of species being over 6 mg L^{-1} (table 2, fig 1B), well above the maximum recorded concentration on the

Oenpelli floodplain of 2.05 mg L^{-1}. The concentration at which chronic, indirect effects were observed for chironomids in a mesocosm experiment (0.2 mg L^{-1}, Temple et al 1991) is displayed on the *x* axis of figure 1B. The environmental concentrations of tebuthiuron exceed this concentration approximately 15% of the time, suggesting the possibility of indirect effects to aquatic invertebrates. The risk of acute effects to freshwater vertebrate species (fish and amphibians) is extremely low and of little concern (table 2, fig 1B). From the available data, acute effects to fish are unlikely to occur below 100 mg L^{-1} tebuthiuron, levels that would not occur in the aquatic environment as a result of Mimosa treatment.

Uncertainty

A number of factors contributed to uncertainty in the effects characterisation. Amongst these were the use of single species laboratory toxicity tests to predict population-level impacts in the natural environment, the limited number of toxicity data points, a lack of knowledge regarding indirect effects of tebuthiuron and the capacity of species to recover following tebuthiuron application, and the influence of confounding factors and stressors.

Uncertainty in the exposure characterisation was exacerbated by the fact that the environmental data originated from only two tebuthiuron applications, both of which were at the same site. Thus, the influence of different environmental conditions in other areas (eg soil type, temperature, soil moisture) on the fate of tebuthiuron could not be fully addressed. In addition, the assumption that dissolved tebuthiuron represented the only bioavailable fraction was not tested.

5 Management issues

Ultimately, the need to reduce the ecological risks of tebuthiuron will be determined by the wider community. Stakeholders may be willing to accept some detriment to wetland biota as a result of tebuthiuron application if the outcome is containment and/or eradication of Mimosa from the area. While this is probably the most ecologically and economically sensible position to adopt, it should be noted that effective and ongoing management plans must initially be in place for Mimosa control in order for the benefits of its eradication to be realised and out-weigh the potential ecological costs of herbicide application.

The efficacy of tebuthiuron has been questioned on several occasions (Cook 1996, Lane et al 1997), and this must also be considered when determining management options. Related to this, there is also a need to determine and compare the ecological risks and efficacy of alternative herbicides, such as metsulfuron and fluroxypyr. This would allow their usage to be managed to reduce the overall risks to the wetland habitats whilst retaining maximum efficacy for Mimosa control.

6 Conclusions

The risk assessment concluded that tebuthiuron represents a signficant and prolonged risk to native freshwater plant species, particularly phytoplankton and floating macrophytes, while the risks to freshwater invertebrates and vertebrates appear low. Although of concern, the overall ecological risks of tebuthiuron (and possibly other herbicides) are probably out-weighed by the known ecological and economic impacts caused by its target weed, *M. pigra*.

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Ecological risk assessment of the cane toad, *Bufo marinus*, in Kakadu National Park[1]

RA van Dam[2], D Walden & G Begg

Background

Cane toads (*Bufo marinus*) entered the Northern Territory (NT) in 1980 from Queensland (Freeland & Martin 1985) and by July 2000 were reported in the upper Mann River and Snowdrop Creek, approximately 15–30 km to the east of Kakadu National Park (KNP) (see van Dam et al 2000, fig 3). Concern about the invasion of cane toads in KNP has been highlighted on a number of occasions, and in 1998 participants at a workshop on the potential impacts and control of cane toads in KNP conceded that a strategic approach for assessing and possibly minimising cane toad impacts should be developed. The first stage would be a preliminary ecological risk assessment to predict the likely impacts of cane toads in KNP and identify key vulnerable habitats and species, with the information being used to develop new, and assess existing, monitoring programs. This assessment, which was conducted by ***eriss*** (van Dam et al 2000) and co-funded by Parks North, addressed potential ecological impacts, whilst also overviewing the potential economic and cultural impacts. This paper focuses on the potential risks to predator species, whilst summarising other potential impacts.

The wetland risk assessment framework developed by ***eriss*** for the Ramsar Convention (van Dam et al 1999) was used to predict key habitats and the species most at risk, in order to provide recommendations for monitoring, and provide a basis upon which Parks North could determine and prioritise management actions.

The risk assessment was based on information from published and unpublished, scientific and anecdotal reports. Information on KNP was derived from relevant research projects undertaken in the Park since the early 1980s. Relevant Territory and Commonwealth agencies were consulted, as were relevant cane toad, native fauna and/or wildlife management experts from around Australia. Discussions were held with community members in the Borroloola and Mataranka regions to gain an indigenous/cultural perspective of the cane toad issue.

Identification of the problem

Since their introduction to Australia in 1935 to control sugar cane pests in Queensland (Mungomery 1935), cane toads have spread naturally and with human assistance throughout much of Queensland, northern NSW and the Top End of the NT (Covacevich & Archer 1975, Easteal et al 1985, Freeland & Martin 1985). The main concern with cane toads is their highly toxic chemical predator defences, with many experimental and anecdotal reports of deaths of

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[1] More detailed discussion of this research is provided in van Dam RA, Walden DJ & Begg GW 2002. *A preliminary risk assessment of cane toads in Kakadu National Park*. Supervising Scientist Report 164, Supervising Scientist, Darwin.

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native predators attempting to consume cane toads (Burnett 1997, Covacevich & Archer 1975, Crossland 1997, Crossland & Alford 1998). The extent of the effect that cane toads have on predator populations in the long term remains controversial, as there is scant published information on this topic. While it is generally acknowledged that a variety of predators will die from mouthing or ingesting toads, whether or not this causes long-term population decline of the predator remains unclear.

The cane toad will soon arrive in KNP[1], a World Heritage area with Ramsar listed wetlands, and high biological diversity, including a large number of rare and endemic species (Press et al 1995). There is serious concern that this particular value of KNP could be diminished if populations of predator species were adversely affected by cane toads. A simple conceptual model of the cane toad life stages that could potentially impact various groups of predators in KNP is shown in table 1.

Table 1 Conceptual model of cane toad life stages potentially impacting upon predator species in Kakadu National Park

Predator group potentially at risk	Life history stage			
	Egg	Larva (Tadpole)	Metamorphling/ Juvenile	Adult
Freshwater invertebrates	*	*		
Fish	*	*		
Amphibia	*	*	*	*
Reptiles		*	*	*
Birds			*	*
Mammals			*	*

Potential extent of cane toads in KNP

Cane toads are likely to colonise almost every habitat type within KNP. The evidence from range expansion of cane toads over the last ten years indicates that most wetland habitats are probably suitable as breeding habitat and also as Dry season refuges (van Dam et al 2000).

The main dispersal within KNP will probably be through the major roads, rivers and streams. Dispersal rates within a catchment could be up to 100 km y^{-1}. The current location of cane toads would indicate an initial progression down the South Alligator catchment via its tributaries (eg Jim Jim Creek, Deaf Adder Creek). Invasion of other areas of the Park will likely depend on which waterways' headwaters are colonised first (eg Mary River, East Alligator River).

Maximum population densities of various cane toad life stages for limited areas of suitable habitat in KNP could be expected to be in the order of: 4000 to 36 000 eggs per metre of shoreline; ~15 to 60 m^{-2} for tadpoles; 2.5 m^{-2} for metamorphlings; and up to 2000 ha^{-1} for adults. However, these will fluctuate substantially depending on temporal and spatial factors.

[1] Note that cane toads had already arrived in Kakadu National Park at the time of publication of this paper.

Potential effects upon predator species

The available anecdotal and experimental information was used to predict the susceptibility of predator species in KNP to cane toads. The degree of susceptibility of cane toad predator species was determined using three criteria:

Definite: documented adverse effects to populations of this species have been reported in the literature;

Probable: documented as having eaten cane toads or their early life stages and effects on individuals reported, but not on populations;

Possible: documented as eating, or thought likely to eat, native frogs or their early life stages, but effects of eating cane toads unknown.

A total of 152 species or species groups were identified under these criteria, covering a broad taxonomic range. Eleven species were considered *definitely* susceptible to cane toads, ie 5 lizard, 3 snake and 3 mammal species. Sixteen species or species groups were considered *probably* susceptible to cane toads, while 125 species or species groups were considered *possibly* susceptible to cane toads.

There are a number of species that are potentially capable of feeding on cane toads without experiencing adverse effects. Some of these species appear relatively immune to the toad's toxin, while others feed on cane toads from the ventral surface, thus avoiding the major concentrations of toxin (Freeland 1990). These species include: some freshwater crustaceans such as prawns and crabs (Crossland unpublished data); the keelback snake (Covacevich & Archer 1975); some species of turtle (Crossland & Alford 1998); several species of birds; (Covacevich & Archer 1975, Freeland 1990); and the water rat (Covacevich & Archer 1975).

Identification of the risks

The data on cane toad effects, distribution and densities are mostly inconclusive and/or show great variability. In addition, information on KNP native species abundance and distributions are deficient. Nevertheless, it is still possible to identify key habitats and also prioritise particular predator species based on (i) the likelihood that they will be at real risk from cane toads, and (ii) their importance to the ecological and/or cultural values of KNP.

Key habitats

As the Dry season progresses, there will be a retreat of cane toads from sites of temporary water to permanent water. The floodplains and sheltered habitats on the margins of floodplains and temporary or shallow billabongs will provide ideal cane toad habitat during the early-mid Dry season. The late Dry season will see cane toads congregate near permanent water or moisture, including permanent billabongs and patches of monsoon rainforest. Few toads would be present in the drier areas of the tall, open eucalypt forest and woodland habitats of the lowland plains.

The Wet season will probably see the highest numbers of cane toad metamorphlings, mainly around the moist margins of the water bodies from which they have emerged. Wet season inundation of the major wetland habitats will see the majority of adult cane toads dispersing into the woodlands and open forests of the lowland plains. The vegetation within the woodlands will provide suitable shelter for cane toads during the Wet season.

Predator species at risk

Predator species were assigned to one of four risk categories, adapted from the original susceptibility criteria (listed above), with associated priority ratings in each category (table 2). The level of risk to, and priority of, a species was assigned using the susceptibility results, and available information on species habitat preferences and feeding ecology. In addition, information on the status of species (ie species listed as endangered, vulnerable, notable or 'flagship' species of KNP) was also used to assign priorities within risk categories.

Table 2 Criteria for determining predatory species most at risk from cane toads

Risk	Priority	Criteria
1. Likely *Population level effects likely*	Highest	Endangered, vulnerable, notable or flagship species considered *definitely* susceptible to cane toads, regardless of relevant habitat information.
	High	As above, but for species not listed as notable or flagship.
2. Possible *Individual mortalities probable, population level effects unknown but possible*	High	Endangered, vulnerable, notable or flagship species considered *probably* susceptible to cane toads, unless relevant habitat/ecological information suggests they are at less risk.
	Moderate	As above, but for species not listed as notable or flagship.; Species considered *possibly* susceptible to cane toads, where relevant habitat/ecological information suggest they are at greater risk.
3. Uncertain *May or may not eat cane toads, with effects on individuals or populations unknown*	High	Endangered, vulnerable, notable or flagship species considered *possibly* susceptible to cane toads, unless relevant habitat/ecological information suggests they are at less risk.
	Moderate	As above, but for species or species groups not listed as notable or flagship; Species considered *probably* susceptible to cane toads, where relevant habitat/ecological information suggests they are at less risk.
4. Unlikely *Effects on individuals or populations unlikely*	Low	Species considered *possibly* susceptible to cane toads, where relevant habitat/ecological information suggests they are at less risk.

A total of 10 species were in risk category one (ie likely effects to populations), the northern quoll being assigned the highest priority due to its listing as notable (Roeger & Russell-Smith 1995). The 9 remaining species, including 5 lizards (all varanids), 3 snakes (all elapids) and one mammal (dingo) were assigned high priority.

Of the 12 species or species groups in the second risk category, none were listed as endangered or vulnerable, or thought to be notable or flagship species, and all species were assigned moderate priority status. Represented in this category were 2 groups of aquatic invertebrates, 3 frogs, one lizard, 3 snakes, freshwater crocodile and 2 birds.

Due to a lack of information, the risk of population level effects was considered to be uncertain for 98 species or species groups, although 21 of these were assigned high priority. These included 3 fish, 3 frog, 3 reptile, 4 bird and 4 mammal species. One of the mammals, the ghost bat, is listed as vulnerable under the EPBC Act of 1999. Given the well documented susceptibility of varanid lizards to cane toads (Burnett 1997), all the varanids within this risk category (two of which are notable) have also been assigned high priority. The remaining 77 species in this risk category were assigned moderate priority and included two groups of invertebrates, 4 fish, 17 frogs, 9 snakes, 42 birds and 3 mammals.

A total of 32 species were considered unlikely to be at risk of experiencing population level effects (based on relevant ecological, feeding or behavioural information), and thus, all were

assigned low priority. These included 12 fish and 18 birds. There is strong evidence to suggest that many fish species are able to detect the noxiousness of cane toad eggs and tadpoles, and avoid eating them (Crossland & Alford 1998, Hearnden 1991). Two non-native mammals, the feral cat and feral pig, while at possible risk, were actually included in this low priority list given their adverse impact on KNP.

Other potential impacts

Quantitative data on impacts to cane toad prey species are scant, and very little could be concluded about the species or species groups at risk. However, termites, beetles and ants constitute the majority of dietary items of cane toads (Begg et al 2000, van Beurden 1980, Zug et al 1975), and as such, these prey groups are the most likely to be impacted, if at all.

Similarly, risks to potential competitor species were unclear. Some native frog tadpoles may be at risk through competition with cane toad tadpoles (eg *L. ornatus*; Crossland 1997). Although adult native frogs do not appear to compete with cane toads (Freeland & Kerin 1988), the potential risk to native tadpoles may impact upon native frog populations.

The major impact upon Aboriginal communities within KNP is likely to be a decline in some traditional foods, and in some situations, the alteration of ceremonies following declines of food and totem species. Cane toads will congregate in areas of human habitation within KNP, will be of nuisance value in these places, and will also represent a risk to domestic and semi-domestic dogs.

Tourism, the major economic activity of KNP, appears not to be at risk from the presence of cane toads.

Uncertainty and information gaps

Major information gaps contributed to the high degree of uncertainty regarding the potential extent and impacts of cane toads in KNP. These include: uncertainty about densities of cane toads in KNP, effects of fire and burning regimes, degree of land/habitat disturbance and the extent to which the Arnhem Land escarpment and plateau will act as a barrier and/or be colonised; the lack of quantitative data on the impacts on animal populations, particularly in the long-term, quantitative data on (pre-impact) KNP faunal populations and distributions as well as dietary information on native species; incomplete knowledge of KNP's invertebrate fauna, many being undescribed and possibly endemic; unknown response and susceptibility of most KNP fish species; unknown competitive interactions with native frogs and other taxa; unknown chemoreceptive response in snakes and their ability to detect cane toad toxins; conflicting and unclear information on freshwater turtles; insufficient information on conservation listed species such as the red goshawk; the lack of experimental or anecdotal evidence regarding effects on bats; and impacts to unidentified endemic species.

Recommendations for additional surveys and monitoring

Priority species for monitoring

Monitoring programs are recommended for all species assigned to risk category 1 (likely). Monitoring of species assigned to risk category 2 (possible) and those assigned high priority in risk category 3 (uncertain) should also be given serious consideration.

Species of particular importance (based on risk, listing as vulnerable or notable, and importance to Aboriginal people) include: northern quoll and some other small mammals (eg sandstone antechinus, red-cheeked dunnart, brush-tailed phascogale); dingo; all the varanid lizards; the northern death adder, king brown snake and western brown snake; the ghost bat; black-necked stork and comb-crested jacana; Oenpelli python; and freshwater crocodile.

Species assigned moderate priority in risk category 3 were not considered priority species for monitoring. However, most of these species were assigned as such due to a lack of information about effects of cane toads. Thus, the risk is considered to be unknown rather than low, and further specific information on these species may result in their re-prioritisation. Monitoring for species assigned to risk category 4 (unlikely) was considered less important.

Priorities for addressing information gaps

A number of information gaps require addressing before more confident estimates of risks can be derived. Monitoring programs assessing the effects of cane toads to KNP species will allow greater understanding of the risks. There is a need for appropriate baseline data, not just for cane toads but for other issues that will arise in the future. In addition, surveys should be conducted to characterise the endemic species of KNP, particularly in the sandstone escarpment/plateau regions. All survey and/or monitoring programs should concurrently monitor cane toad abundances and habitat preferences. Other information gaps that could be addressed, but are less of a priority, include the effects of fire on cane toads, and information lacking for particular species or species groups (eg freshwater turtles, red goshawk).

Risk management and reduction

Parks North have initiated a cane toad identification training program and rapid response strategy to manage human-assisted incursions of cane toads. Additionally, frog recording stations are continuing to be established at sites in KNP. Baseline data have been collected for the past two wet seasons.

Very little can be done to reduce cane toad numbers in KNP. Particular measures may prove effective in localised areas (eg townships, caravan parks), but efforts would need to be sustained. Construction of physical barriers around sites may not be relevant to Park management. Management of feral pig damage may help reduce the densities of cane toads in pig-affected areas. Chemical and biological control methods are insufficiently developed at this stage.

It is recommended that the invasion of cane toads be managed initially by i) ensuring that monitoring efforts are underway to assess the impacts of cane toads upon the values of KNP, and ii) investigating measures by which cane toads can be managed on a localised basis.

The preliminary ecological risk assessment (van Dam et al 2000) provided a starting point from which to determine the monitoring requirements for fauna. In addition, although not addressed here, it has provided an overview of the potential cultural and socio-economic impacts, which could be studied in greater detail by appropriate experts.

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Vulnerability assessment of two major wetlands in the Asia-Pacific region to climate change and sea level rise[1]

RA van Dam[2], A Mapalo[3], L Peiying[4], CM Finlayson & D Watkins[5]

Introduction

Given the importance of coastal wetland habitats in the Asia-Pacific region to both people and for biodiversity, and the potential for these to be impacted by climate change and sea level rise, vulnerability assessments of two major wetlands in the region were undertaken. The sites chosen were the Yellow River Delta (YRD) in China, and Olango Island in the Philippines. These have recognisable value for both people and for biodiversity, with both sites being listed under the East-Asian Australasian Shorebird Reserve Network, and Olango Island also being listed as a wetland of international importance under the Ramsar Wetland Convention.

The study's major objectives were to raise awareness of the issue of climate change and sea level rise in the Asia-Pacific region, to provide advice and training to national and local agencies on procedures for vulnerability assessment, and specifically, to obtain a preliminary understanding of the potential impacts of climate change and sea level rise on the biological, physical and socio-economic attributes of the two wetland sites.

The assessments were based on the model provided by Bayliss et al (1997) using a procedure presented by Kay and Waterman (1993) and Waterman (1995), and included the following steps:

- description of the physical, biological and socio-economic attributes of the site;
- development of a predicted climate change scenario based on existing literature;
- identification of existing natural and anthropogenic 'forcing factors' and their impacts;
- assessment of vulnerability to existing forcing factors;
- assessment of vulnerability to climate change and sea level rise;
- documentation of current responses to coastal hazards;
- recommendations for future monitoring requirements and management strategies;
- identification of information gaps and research priorities.

The following overviews of the vulnerability assessments for the YRD and Olango Island are summarised from Peiying et al (1999) and Mapalo (1999), respectively.

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[1] More detailed discussion of this research is available in van Dam RA, Finlayson CM & Watkins D (eds) 1999. *Vulnerability assessment of major wetlands in the Asia-Pacific region to climate change and sea level rise.* Supervising Scientist Report 149, Supervising Scientist, Canberra.

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Yellow River Delta

The YRD (fig 1) was chosen as a study site primarily because it has been nominated for the East Asian–Australasian Shorebird Reserve Network. Due to its importance as a habitat for migratory and resident shorebirds (Barter et al 1998), a 1500 km^2 Nature Reserve has been established along the eastern coast of the delta.

The YRD represents the meeting point of the Yellow River with the Bohai Sea, in eastern China (Cheng 1987). The delta covers approximately 6000 km^2, although historically it has been in a dynamic state due to the high sediment load and frequently changing course of the Yellow River (Cheng 1987; see fig 1b). More recently, the river course has been stabilised, allowing substantial development to occur. The YRD is now a highly urbanised and industrialised region, with a population of 1.64 million and major industries including oil extraction and crop and cattle farming (Wang et al 1997a). Subsequent demands on water resources, both from within and upstream of the YRD have greatly reduced the flow of the Yellow River in the last decade (Lu et al 1997). The Nature Reserve was established in recognition of the YRD's importance as a site for migratory and non-migratory shorebirds (Barter et al 1998). However, it is under great pressure from urbanisation, farming and oil and natural gas extraction.

The major physical attributes of the YRD include the river and underground water, the low topographical relief of the delta, the geomorphic units of the terrestrial delta, the subaqueous delta and the tidal flats, the sediment load of the Yellow River and subsequent sedimentation (Gao & Li 1989, Li et al 1992, Xu et al 1997, Yang & Wang 1993, Zang et al 1996), and the natural resources of oil, gas and water (Wang et al 1997a). The major biological attributes include terrestrial and aquatic plants and animals, particularly the birdlife, which includes 152 species of protected birds (Xu et al 1997, Zhao & Song 1995). Over 500 000 shorebirds are estimated to utilise the wetlands of the YRD during their northward migration (Barter et al 1998).

The predicted climate change scenario for the YRD was based on regional climate change scenarios for temperate Asia or China specifically, by the Intergovernmental Panel on Climate Change (IPCC) and other investigators. The scenario used for this study included the following estimates:

- A rise in relative sea level of 48 cm by 2050 (specific for the YRD; Chen et al 1997);
- A rise in mean air temperature of 1.4°C by 2050 and 3°C by 2100 (for China/East Asia, Hulme 1992);
- A rise in annual precipitation of 2–4.5% by 2050 (for East China, Wang & Zhao 1995).

The major natural forcing factors acting on the YRD (excluding climate change) are sedimentation, the Asian monsoon, El Niño, flooding and storm surge. Major impacts associated with these include erosion and expansion of the coastal wetlands, damage to infrastructure, crops and livestock, and loss of human life (Chen et al 1997, Science & Technical Committee of Shandong Province 1991, Lu et al 1997, Mo et al 1995, Song et al 1997). Major anthropogenic forcing factors include the large population and associated types of land use, oil and natural gas development, and water and air pollution. The major impacts include a reduction in freshwater supply, a reduction in surface and groundwater quality, degradation of the Nature Reserve and the subsequent loss of wetland habitat and biodiversity (Wang et al 1997b,c).

The YRD is already extremely vulnerable to existing forcing factors. Although river flows have decreased in the last decade, the YRD is still highly vulnerable to flooding from both upstream sources and from storm surges. The high utilisation of water resources, while aiding in the development of industry and agriculture and enhancing the standard of living, will eventually result in major ecological consequences, such as salinisation, loss of wetland habitat and desertification. Without proper management, urban, industrial and agricultural activities will further pollute the already poor quality waters within the YRD.

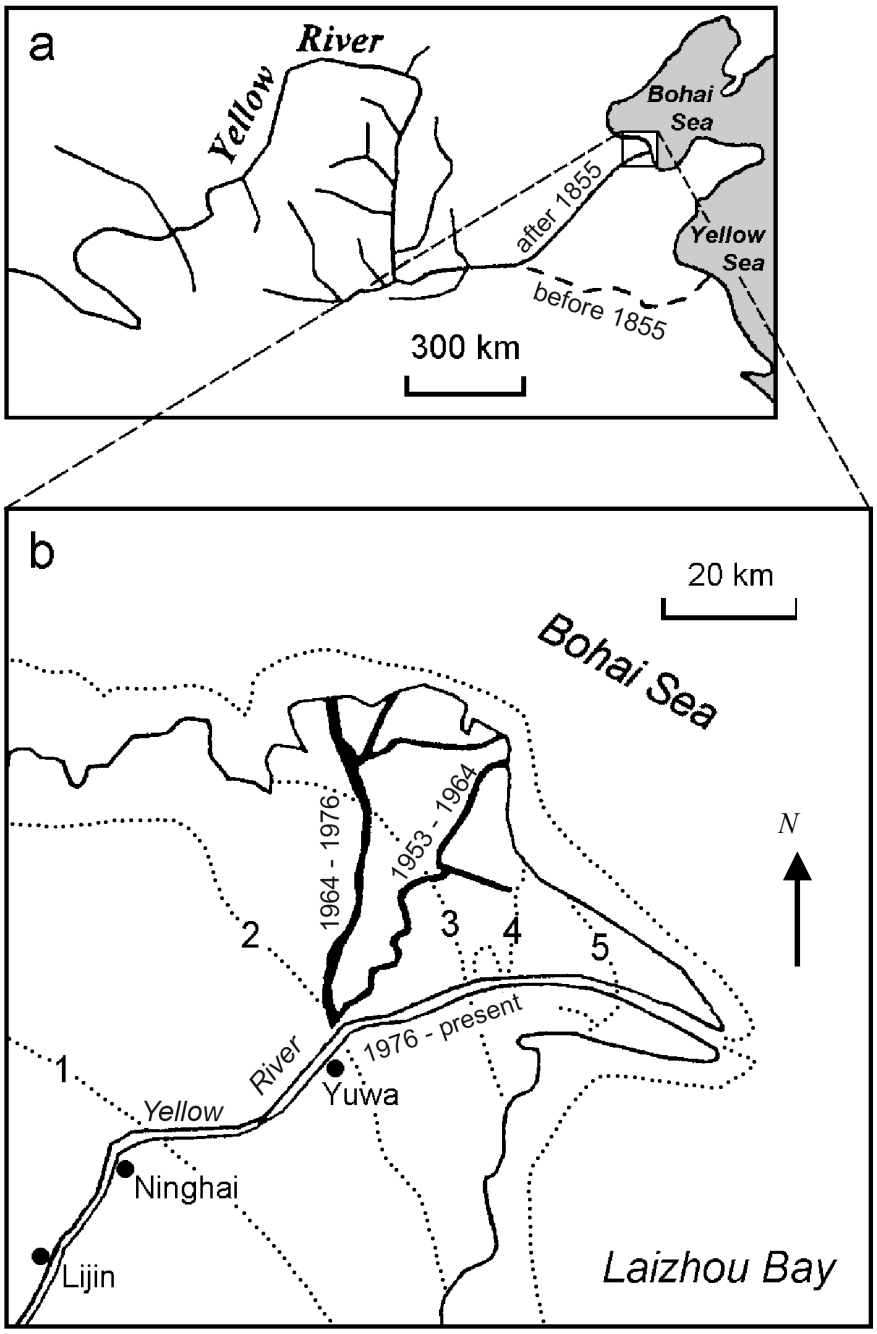

Figure 1 Map of (a) the location of the Yellow River basin, and (b) the modern Yellow River delta with the historical changes of the coastline: 1. coastline of 6000 years BP; 2. coastline of 1855; 3. coastline of 1934; 4. coastline of 1976; 5. coastline of 1980

The YRD is also vulnerable to predicted climate change and sea level rise. Increased moisture stress, insect pests and plant diseases resulting from climate warming are expected to have unfavourable effects on agricultural production. Salt marshes and other coastal wetlands are thought to be particularly vulnerable to permanent inundation and erosion as a result of sea level rise and increased storm surge. This would have flow-on effects to tourism, freshwater supplies, fisheries and biodiversity. Sea level rise will result in a number of other impacts including a reduction in the protective capacity of the dyke systems. Assuming a 1 m sea level rise and 2–3 m storm surge, approximately 40% of the YRD could be inundated. Saltwater intrusion will also be a major issue, further reducing already limited freshwater resources. The above impacts will have major consequences for both the socio-economic and biological attributes of the YRD.

A series of dyke systems have been in place in the YRD for many years to protect against floods both from upstream and from storm surges (Lu et al 1997, Zhang et al 1997). Some of these have been upgraded whilst others require attention. Many of these flood control dykes will serve as protective barriers to sea level rise and increased storm surge, although the extent to which they can protect the adjacent land is uncertain. Other control measures are in place to prevent or minimise floods resulting from ice jam in the river (Lu et al 1997, Zhang et al 1997). Freshwater shortages are being addressed by increasing the capacity of existing reservoirs or proposing the construction of new reservoirs.

The study identified a number of management strategies or countermeasures for protecting the YRD from both existing forcing factors and predicted climate change and sea level rise including:

- Integration of information from programs monitoring sea level rise, coastal zone ecology and sensitivity, and socio-economic and cultural indicators;
- Stabilisation of the course and mouth of the Yellow River;
- Consideration of flood risk in urban and industrial planning;
- Protection and management of coastal wetlands and the Nature Reserve;
- Control of urban and industrial pollution;
- Establishment of reservoirs for water storage and conservation; and
- Increasing community awareness about environmental protection.

In addition, recommendations regarding the management of the Nature Reserve included:

- Development of an appropriate administrative and management system;
- Drafting and implementation of appropriate environmental protection laws;
- Increasing scientific research to provide a basis for management; and
- Enhancing community awareness of ecology and environmental protection.

The YRD currently faces a range of serious ecological and socio-economic problems, most of which are related to water supply, be it in shortage, excess (flooding) or of poor quality. These issues highlight the need to consider both economic development and environmental protection when planning the future sustainable development of the YRD. In addition, it is now imperative that the issue of climate change and sea level rise is incorporated in any such plans. This study highlights the vulnerability of the YRD to predicted climate change and sea level rise, particularly in terms of exacerbating the region's current water supply and quality problems. The proposed management strategies provide the first step in effectively addressing the issue of climate change and sea level rise.

Olango Island

Olango Island (fig 2) was chosen as a study site for several reasons. It is a small, coral reef island (~6 × 3 km) with low topographical relief and a maximum elevation above sea level of only 9 m, it sustains a population of over 20 000 and is already under pressure from anthropogenic activities including fishing, groundwater extraction and mangrove harvesting; it is a major wetland site for shorebirds, being nominated for the East Asian–Australasian Shorebird Reserve Network and listed as a wetland of international importance by the Ramsar Wetland Convention (CRMP 1998). Due to its importance as a flyway stopover site, a 920 ha wildlife sanctuary was established in the south of the island (DENR 1995).

The major physical attributes of Olango Island include the low topographical relief, sandy shorelines and limestone outcroppings, the groundwater lens and the monsoonal climate (CRMP 1998, DENR 1995, Ligterink 1988, PAGASA 1998). The major biological attributes include mangrove forests, seagrass beds, coral reefs, birdlife and other wetland fauna (CRMP 1998, Davies et al 1990, DENR 1995, Magsalay et al 1989, Paras et al 1998, SUML 1997). The major socio-economic attributes include the large population in general, livelihood activities such as fishing and shell and seaweed collection, infrastructure and freshwater supply (CRMP 1998, Ligterink 1988, Remedio & Olofson 1988, SUML 1997, Walag et al 1988).

The predicted climate change scenario for Olango Island was based on predicted regional scenarios by the IPCC and the Philippine Atmospherical, Geophysical and Astronomical Services Administration (PAGASA) where possible. Where such information did not exist, estimates from IPCC global scenarios were used.

The predicted scenario for Olango Island was:

- A rise in mean sea level of 30 cm by 2030, and 95 cm by 2100 (Watson et al 1996);
- An increase in mean global sea surface temperature of 0.5°C by 2010 and 3°C by 2070 (Whetton et al 1994);
- A 20% increase in typhoon intensity (Henderson-Sellars & Zhang 1997);
- A tendency for increased rainfall, intensity and frequency (Whetton et al 1994).

The major natural forcing factors on Olango Island are the south-west and north-east monsoons, typhoons, storm surge and El Niño. Some of these have positive impacts on the island, by way of recharging the underground water supply, while the major negative impacts include flooding, erosion and infrastructure damage (Bagalihog & Redentor 1996, CRMP 1998). The major anthropogenic forcing factors involve the exploitation of natural resources, such as over-fishing and illegal fishing, over-extraction of groundwater, mangrove harvesting and coral extraction (CRMP 1998). These factors could result in erosion, saltwater intrusion, shortages of freshwater, habitat destruction and the loss of biodiversity.

Assessment of the vulnerability of Olango Island to existing forcing factors indicated that the island is already under enormous pressure, mostly from natural resource exploitation, although typhoons and associated storm surges also exert negative impacts. Many of the natural resources are already severely degraded, particularly the fisheries and the under ground supply of freshwater. The sustainability of these resources is in doubt, although recent management recommendations have provided the first step towards long-term sustainability.

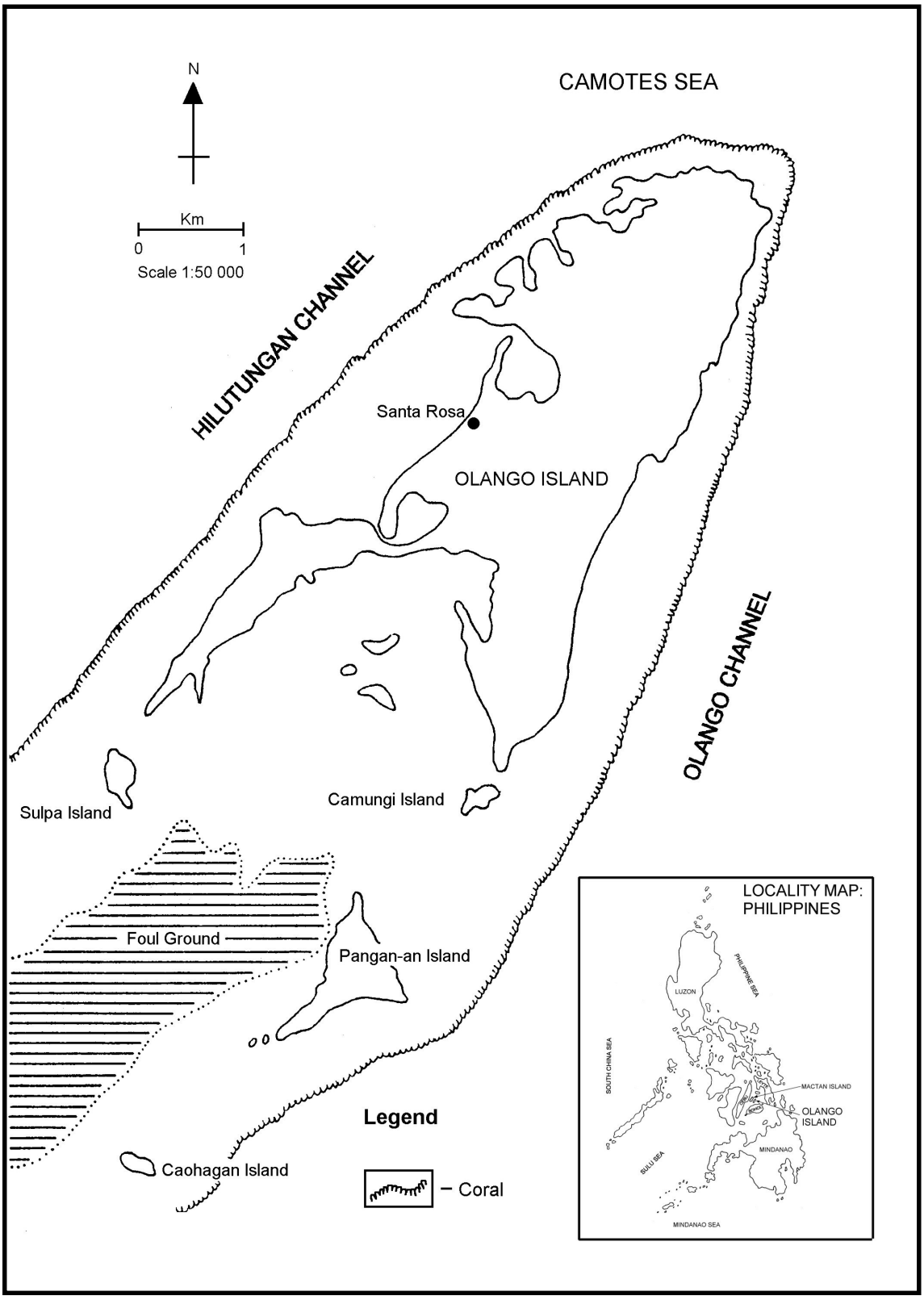

Figure 2 Map of Olango Island showing the major geographical features

Climate change and sea level rise will undoubtedly place additional stress on Olango Island and its attributes. Given its low elevation and topographical relief, more than 10% of the current land mass would be lost in the event of a 95 cm rise in sea level. In addition, more severe typhoons and storms surges would result in an even greater portion of the island being subjected

to inundation and flooding. Given that the majority of human settlement on the island occurs in close proximity to the shoreline, this represents a major problem. An increase in sea level would also facilitate saltwater intrusion into the underground freshwater lens, although this could be offset by an increase in rainfall. Potential effects on the biological attributes include loss of mangrove stands due to an inability to recolonise inland, bleaching and death of corals due to increased sea surface temperature, and loss of feeding grounds and roosting habitat for resident and migratory shorebirds. Potential effects on socio-economic attributes include the displacement of people, loss of infrastructure and loss of livelihood options.

While the current issues facing Olango Island are immediate and serious, the vulnerability of the island to climate change and sea level rise is sufficiently great to require consideration in future management plans.

Current responses to the current and future hazards facing Olango Island include a number of resolutions and ordinances at the local (Barangay) level, such as the declaration of local fish sanctuaries and marine reserves, and prohibition of sand extraction and illegal fishing (CRMP 1998). Regional responses, such as the Mactan Integrated Master Plan (Lapulapu City 1996) address land use issues for Olango Island, while DENR has drafted management recommendations for the wildlife sanctuary, in which the issue of climate change and sea level rise is recognised (DENR 1998). DENR also conducts a bird monitoring program in the wildlife sanctuary. The USAID-funded Coastal Resource Management Project (CRMP) has completed a Coastal Environmental Profile of Olango Island, which will assist in developing a coastal zone management plan (CRMP 1998). On a national scale there also exist a number of plans and policies relating to coastal zone management and mitigation/protection plans against coastal hazards.

Major parameters recommended for future monitoring included: geophysical parameters such as storm surge, shoreline erosion, mean sea level, groundwater salinity and water and air temperature; biological parameters such as bird populations, mangrove growth and distribution, seagrass cover, coral cover and reef fish biomass; socio-economic parameters such as tourism growth, population structure and infrastructure development. A number of future management strategies are also proposed, including the creation and maintenance of buffer zones, the provision of livelihood opportunities for the local people and developing awareness of techniques for natural resource management. Management measures to address potential impacts of climate change and sea level rise include reviewing the feasibility of physical barriers to protect against storm surge, prohibition of shoreline vegetation harvesting, regulation of groundwater extraction, protection of the groundwater catchment area, establishing fish sanctuaries, seeking alternative livelihoods, developing a formal education program and reassessing future coastal development plans.

A number of information and research gaps were also identified. There were major deficiencies in storm surge data, the quantification of coral and sand extraction, natural disaster damage estimates for lives, property, and natural resources, groundwater salinity and transmissibility data, the biology and ecology of endangered species, and the impacts of mangrove forestation on the seagrass beds. In addition, the lack of a detailed topographic map made it difficult to make precise estimates of the potential impacts of sea level rise on the island.

The vulnerability assessment highlighted the magnitude of the immediate threats facing the local communities and natural resources of Olango Island. First and foremost among these threats are the increasing population and the associated depletion of the fisheries and underground freshwater supply. Even in the absence of climate change and sea level rise, sustainability of these resources will not be achievable if management plans do not address

the problems. Olango Island possesses many characteristics that make it highly vulnerable to climate change and sea level rise; it is a small, low-lying coral reef island with a large, technologically poor population. Thus, climate change and sea level rise will only serve to place further stress on those natural resources that are already under threat. Subsequently, recently drafted local, regional and national management plans need to recognise and address the possible consequences of climate change and sea level rise.

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Information for a risk assessment and management of *Mimosa pigra* in Tram Chim National Park, Viet Nam

D Walden, CM Finlayson, R van Dam[1] & M Storrs[2]

Introduction

Tropical wetlands are renowned for providing many values and benefits for people and for supporting a diverse and plentiful biota (Finlayson & Moser 1991, Dugan 1993). There is also increasing pressure on such wetlands as human populations increase and development activities affect the wetlands and their catchments. Responses to such pressures have varied and, as a consequence, many wetlands have been lost and degraded. This is the situation that exists in Viet Nam where the wetlands in Tram Chim National Park represent but a remnant of the habitats that existed some 25 years ago (J Barzen pers comm 1999).

Within this context we have collated an information base on the biology and management of *Mimosa pigra* (known colloquially as mimosa) as a case study for the application of a formal risk assessment procedure designed to assist weed managers in Viet Nam (and elsewhere). Much of the information for this assessment has come from northern Australia where mimosa has been seen as a major weed for more than two decades. Mimosa has increasingly become a major menace in South East Asia (Lonsdale 1992) and is a constant menace to both food production and nature conservation.

Wetland risk assessment

Over the last decade the concept of environmental risk assessment developed and expanded from a narrow and precise analysis of quantitative ecotoxicological data to more general and qualitative analyses of environmental problems. This led to development of a generic model for wetland risk assessment coupled with advice on the deployment of early warning systems for detecting adverse ecological change in wetlands (van Dam et al 1998). The model provides guidance for environmental managers and researchers to collate and assess relevant information and to use this as a basis for management decisions that will not result in adverse change to the ecological character of the wetland.

The six steps in this model are: i) identification of the problem (eg site assessment; site-specific information); ii) identification of the effects (eg field assessment by surveys or surveillance); iii) identification of the extent of exposure (eg level of infestation or concentration); iv) identification of the risk (comparison of the field surveys with extent of infestations); v) risk management/risk reduction (implementation of management practices); and vi) monitoring (early warning and rapid assessment techniques).

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Case study — *Mimosa pigra*

The case study has involved an initial step of reviewing the literature and talking with field operators and wetland managers to identify the following: life cycle features of mimosa and its invasive potential; habitat range of mimosa and its likely distribution; ecological effects of mimosa and its likely impact; economic effects of mimosa and its likely impact; and control measures used against mimosa and their likely success.

In undertaking this assessment we have recognised that mimosa is an acknowledged major weed and that control measures are urgently needed. This provides the basis for weed management strategies proposed specifically for use at Tram Chim.

Life cycle of *Mimosa pigra* and its invasive potential

The life cycle and general biology of mimosa have been described in recent years (Lonsdale 1992, Lonsdale et al 1995, Miller 1988, Rea 1998).

Mimosa is native to tropical America where it occurs in a wide belt extending from Mexico through Central America to northern Argentina. It has been introduced to other areas as an ornamental, a cover crop, or for erosion control, and is now widespread and a serious weed in Africa, Asia, some Pacific islands, and most spectacularly in the northern part of the Northern Territory, Australia.

Description

When mature, mimosa is an erect, much branched prickly shrub reaching a height of 3–6 m. Stems are greenish at first but become woody, are up to 3 m long, and have randomly scattered, slightly recurved prickles 5–10 mm long. Leaves are bright green, 20–25 cm long and bipinnate, consisting of about 15 pairs of opposite primary segments 5 cm long with sessile, narrowly lanceolate leaflets that fold together when touched or injured and at night.

The flowers are pink or mauve, small, regular and grouped into globular heads 1–2 cm in diameter. The heads are borne on stalks 2–3 cm long, with two in each leaf axil, while the corolla has four lobes with eight pink stamens. The fruit is a thick hairy, 20–25 seeded, flattened pod borne in groups in the leaf axils, each 6.5–7.5 cm long and 7–10 mm wide. The fruit turns brown when mature, breaking into one-seeded segments. The seeds are brown or olive green, oblong, flattened, 4–6 mm long, and 2 mm wide.

Features promoting survival and dispersal

Mimosa has many features that are generally considered ‘advantageous’ to a weed. It is able to tolerate anaerobic substrates by sprouting adventitious roots that can absorb oxygen. This enables the plant to survive reasonably deep flooding and to advance into deep water habitats. Further, it can resprout from the remaining stem-base if cut or broken. Under some circumstances if burnt, a large proportion of mature plants and about half the seedlings may regrow, probably from dormant buds (Miller & Lonsdale 1992).

The plants mature quickly and can set seed in their first year of growth. The seeds are contained in individual segments of seed-pods that ‘burst’ apart when mature. The segments are covered with bristles that enable them to adhere to animals and clothing, and to float on water for extended periods. The seeds are also dispersed in soil and mud, adhering to vehicles and other machinery (Lonsdale et al 1985). The lifespan of the seeds in the ground depends on the soil type and the depth at which they are buried. For example, half of a seed population

was no longer viable after 99 weeks at a depth of 10 cm in a light clay soil, while a similar loss in viability was observed after only 9 weeks in a heavier cracking clay (Lonsdale et al 1988). In sandy soils the lifespan of seeds may be much longer. Dormancy of seeds in the soil is broken by expansion and contraction of the hard seed-coat by temperature changes ranging from about 25–70°C. Seeds buried deeper than 10 cm generally do not successfully germinate unless brought to the surface.

Seed rate production has been measured between 9000–12 000 per year depending on the conditions (Lonsdale et al 1988). If a mere handful of seeds m^{-2} were to germinate, the resulting plants, with rapid growth rates and early maturation (it takes as little as six months from germination to flowering), could form dense stands and start copious seed production all over again.

Spread of mimosa in northern Australia

Mimosa was probably introduced to the Northern Territory, Australia, at the Darwin Botanic Gardens in the 20 years prior to 1891, either accidentally in seed samples, or intentionally, as a curiosity, because of its sensitive leaves (Miller & Lonsdale 1987). It lingered in the Darwin region causing an occasional nuisance (Miller & Lonsdale 1987) until it was noticed some 95 km to the south near the township of Adelaide River in 1952.

It was further spread by particularly heavy flooding in the 1970s. At this time the floodplains were being overgrazed and trampled by large herds of feral Asiatic water buffalo (*Bubalus bubalis*). Overgrazing removed much of the natural vegetation, reducing competition for the less palatable mimosa. As a result, mimosa seeds were rapidly spread to bare and highly disturbed soils which became ideal seedbeds (Lonsdale & Braithwaite 1988).

In 1975 only a few mimosa plants were known to occur on the Adelaide River floodplain. By 1978 the infestation covered an estimated 200–300 ha with impenetrable thicket; by 1980 there were plants scattered over an estimated 4000 ha (Miller et al 1981); and in 1984 the population was estimated to cover about 30 000 ha in dense and scattered stands (Lonsdale 1993). At some point the plant appeared in other floodplain systems, such as along the Daly, Finniss, Mary and East Alligator rivers. By 1989 mimosa infestations had reportedly increased to 80 000 ha, a figure which has not been substantiated. Unfortunately no contemporary estimate is available.

Habitat range and likely distribution

Mimosa favours a wet-dry tropical climate and has been introduced into most tropical regions of the world where it grows in comparatively open, moist sites such as floodplains, coastal plains and river banks. In the introduced range it readily infests areas that have been disturbed as a consequence of human activities, such as reservoirs, canal and river banks, roadside ditches, agricultural land and overgrazed floodplains. In Australia and Thailand it forms dense thickets covering thousands of hectares (Lonsdale et al 1985, Napometh 1983). In its native range it occupies similar habitats, especially in areas which have been disturbed, but usually occurs as small thickets or as individual plants (Harley 1985). In Costa Rica, part of its natural range, it is common in overgrazed areas (Boucher et al 1983).

In Australia mimosa is apparently not restricted to any one soil type. The relationship between the plant's distribution and salinity levels remains to be determined, although tolerance to higher salinities (ie ~18 ppt) has been observed (Miller 1983).

Ecological effects

Mimosa poses an enormous problem in Australia where a largely 'natural' landscape is being completely altered, with floodplains and swamp forest being invaded by dense monospecific stands of mimosa, which have little understory except for mimosa seedlings and suckers. For native species, the impact of such a change in the habitat is severe. Many animals have become scarce or have disappeared altogether. In general, mimosa thickets support fewer birds and lizards, less herbaceous vegetation, and fewer tree seedlings than native vegetation (Braithwaite et al 1989).

Coverage of wetlands by mimosa could drastically affect waterbird populations, which rely on sedgeland for breeding and feeding. Swamp forests with open canopies, such as those dominated by species of *Melaleuca,* are prone to invasion with the formation of a dense understory that prevents seedlings of the forest trees from establishing. Thickets of mimosa also prevent light penetration to species on the ground (Braithwaite et al 1989).

Some species have increased in number as a result of the presence of mimosa. In northern Australia the most notable of these is a rare marsupial mouse called the red-cheeked dunnart (*Sminthopsis virginiae*) (Braithwaite & Lonsdale 1987). However, small mammals will only benefit where the weed occurs in patches from which they can make forays into the surrounding vegetation for food.

Economic effects

In addition to adversely affecting the natural flora and fauna, mimosa can also interfere with stock watering, irrigation, tourism, recreational use of waterways, and the lifestyles of indigenous peoples. It can smother pastures, reduce available grazing areas and make mustering difficult (Miller et al 1981). In Thailand it has caused sediment to accumulate in irrigation systems and reservoirs, created safety hazards along roads, and made access to electric power lines difficult (Robert 1982, Napometh 1983, Thamasara 1985).

In many cases such economic impacts are contingent with ecological impacts. For example, tourism is affected directly by restricted access to floodplains and other sites, but also by loss of income in a range of associated service activities and can lead to a reduction in the number of visitors. As early as 1981 such effects were felt in northern Australia (Miller et al 1981). Further economic losses could occur in northern Australia if infestations of mimosa restrict access for the recreational fishing industry which has an economic impact amounting to millions of dollars (Julius 1996, Griffin 1996).

The above mentioned impacts of mimosa in northern Australia also affect Aboriginal land use practices. Aboriginal people continue to rely on the natural environment for both their spiritual and physical well being; practices such as hunting and foraging not only provide people with food, but are closely tied to spiritual beliefs and traditional law, and allow each generation to share extensive environmental knowledge with succeeding generations.

Another economic impact is the financial cost of controlling the weed. In northern Australia it is estimated that over A$20 million (approx. US$12 million) has already been spent by government and landholders on research and control of mimosa (M Storrs pers comm 2000).

Control measures

In northern Australia the recommended strategy for controlling mimosa is to prevent initial invasion of the weed, eradicate small infestations by physical or chemical means and, for large infestations adopt an integrated approach involving biological control, herbicide

application, mechanical removal, fire and pasture management. Despite differences in land use practices many aspects of this strategy could be applicable in Viet Nam and elsewhere.

Common problems encountered with controlling mimosa are i) a lack of awareness of the problems that could occur if the weed is not effectively controlled, and ii) discontinuity in control. Interruptions in control programs wastes time, resources and funds, and allows mimosa time to recover from past treatment (Miller et al 1992).

Prevention

Preventative weed control is arguably the most cost efficient form of weed management and can play an integral role in strategic weed management. Part of the preventative approach for mimosa involves comprehensive surveys to identify isolated infestations that should be targeted before they expand and become impossible to control (Cook et al 1996). Preventative measures include educating the community, and placing controls over likely sources of seeds, such as stock feed, soil and sand from infested areas, and restricting the movement and/or cleaning of vehicles and stock that frequent infested areas (Benyasut & Pitt 1992).

Physical and mechanical control

Physical and mechanical methods of weed control have been used extensively and many can be applied using relatively unskilled labour and make use of readily available equipment. However, at best they are only temporary control options for large infestations. Thus, it is recommended that they are used in combination with herbicide application and burning (Miller 1988, Miller et al 1992, Miller & Lonsdale 1992).

Hand weeding

Hand weeding is usually employed on small plants or seedlings and can be very effective for controlling seedlings amongst crops, but may not be practicable when they are present in large numbers or when the plants are large. Seeds should be collected from the plants before weeding and then burnt in a container. Roots should be removed from the soil and, after weeding, the plants should be left out of contact with wet soil to prevent striking.

Hand implements

Hand-hoeing or grubbing with a mattock is faster and more effective than pulling by hand. Again, it is important that the roots are removed. Long handled cutters, axes and machetes may be used to cut plants, however, stumps may quickly resprout, making this a temporary measure only. Regrowth may be stopped by immediate application of a herbicide or by flooding as the stumps will die if submerged for more than 30 days (Thamasara 1985).

Power operated equipment

In areas under cultivation young mimosa seedlings can be controlled by rotary-hoeing and ploughing. Tractors allow large areas to be controlled quickly. Slashing or mowing can be used as a temporary measure, but a heavy duty machine is needed and regrowth may be rapid. Motor-driven cutters and chainsaws are more efficient than hand implements for cutting larger plants.

Ecological control

Use of fire

The use of fire as a control mechanism is limited because the plants have low flammability. Dense thickets will not usually support a fire due to the lack of understory fuel. Further, when

infestations are burnt, fire does not have a major impact on mature plants, although this can vary depending on the season and weather conditions (Miller 1988). Mature plants can sprout quickly. Mortality in seedlings is greater but often still more than 50% regrow after fire.

Fire can have varying effects on mimosa seed, depending on the fuel load and the position of the seed in the soil profile. It can increase seed germination by scarifying the hard seed coat while some of the seed on the surface may be killed, but beneath the soil surface there is only a small rise in temperature, the effect penetrating to about 5 cm.

Use of competitive pastures

Mimosa seedlings are susceptible to competition from grasses. However, control of dense, mature mimosa using competitive pastures alone is unlikely. Pasture management could be most useful in situations of incursion prevention and after the application of herbicides, mechanical control and burning, in particular where the mimosa canopy is opened up to allow either natural regeneration of native species or the sowing of other species to compete with mimosa seedlings (Miller 1988).

Reduction of grazing pressure

Mimosa is opportunistic and will often germinate in areas that have been disturbed by grazing animals or have been denuded by overgrazing. The removal or reduction in grazing pressure is usually important in allowing re-establishment of more desirable species, thus assisting in weed control.

Chemical Control

Herbicides used for control of mimosa

Chemical control has been extensively used in northern Australia and Thailand. Table 1 lists herbicides that have been tested in an attempt to replace 2,4,5-T, which was the main herbicide used in the 1960s and 1970s. Five chemicals that are commonly used today in the Northern Territory are described in table 2.

Application methods

The most effective time to apply herbicides is usually during the period of active growth (for herbicides whose translocation is reduced by inactive growth) and before the plants have produced mature seed, in order to reduce the plant population the following year. For mimosa this is most likely in the early or even mid-Wet season (Lonsdale 1988, Miller 1988). However, due to the height, density and prickly nature of mimosa, access can often be difficult unless aircraft are used. This immediately introduces the potential for herbicide drift to off-target species and contamination of adjacent habitats. The application of pelletised and granulated herbicides can greatly reduce the problem of drift as can applying liquid herbicides during favourable climatic conditions, such as high humidity, and lower temperatures and wind speed (Miller 1988). Ground-based methods of applying herbicides include direct injection, foliar or basal bark spraying, and soil application of both pelletised and liquid herbicides. All have particular advantages and risks and can be expensive.

Table 1 Herbicides and methods of application evaluated for the control of *Mimosa pigra* in Australia and Thailand (from Miller & Siriworakul 1992)

Herbicide	Method of application					
	Soil	Cut stump	Stem injection	Basal bark	Foliar — ground	Foliar— air
Atrazine					*	
Clopyalrid					*	*
Dicamba	*	*	*	*	*	*
Dicamba + MCPA					*	*
Ethidimuron	*					
Fluroxypyr					*	*
Fosamine					*	
Glyphosate		*	*		*	
Hexazinone	*	*	*		*	
Imazapyr		*			*	
Karbutilate	*					
Metsulfuron methyl					*	*
Picloram + 2,4-D			*	*	*	
Picloram + 2,4-D + triclopyr			*	*		
Picloram + 2,4,5-T		*	*	*		
Picloram + triclopyr		*	*	*	*	*
2,4,5-T					*	
Tebuthiuron	*					
Triclopyr		*	*	*	*	

Table 2 Features of herbicides used to control *Mimosa pigra* on Aboriginal land in northern Australia

Chemical	Proposed max rate g/ha a.i.	Mimosa mortality [1]	Control of regrowth [2]	Residual activity [3]	Toxicity [4]	Selectivity [5]	Ease of use [6]
Tebuthiuron	2000	H	H	H	M	H	H
Fluroxypyr	600	M	H	L	M	M	M
Hexazinone	0.8	H	H	M	M	L	M
Metsulfuron	45	H	H	L	L	H	M
Dicamba	1200	L	M	L	M	H	M

1 *Mimosa* mortality assuming optimal conditions: H ≥98%; M = 90–98%; L = 70–90%.

2 Regrowth control assuming typical wetland conditions: H = >6 months; M = 3–6 months; L ≤3 months.

3 Residual activity of herbicide assuming typical wetland conditions: H = >6 months; M = 3–6 months; L ≤3 months.

4 Toxicity based on mammalian toxicity (LD50 mg/kg): M = slightly toxic (500–5000); L = practically non-toxic (5000–15000).

5 Selectivity of herbicide: H = highly selective; M = moderately selective; L = not selective.

6 Ease of use: H = very easy to use; M = easy to use; L = moderately difficult to use.

Monitoring and impacts of herbicides

The application of large amounts of herbicides has been viewed with concern and a number of monitoring and assessment programs have been instigated. The most notable of these in northern Australia was undertaken near Oenpelli (Gunbalanya) some 300 km to the west of Darwin where non-target plant species, such as *Melaleuca* trees and sedges were killed by applications of tebuthiuron (Schultz & Barrow 1995). Whilst the use of these chemicals was accompanied by various environmental measurements they were not preceded by specific toxicological testing using local species. For tebuthiuron this was justified on the basis that an urgent control situation existed and its effects on northern hemisphere temperate species had been extensively studied. Subsequent tests using non-target native species indicated that toxicity to native aquatic animals is very low compared to aquatic plants (Camilleri et al 1998).

Biological control

In 1979, a biological control program was initiated in northern Australia, however, whilst this may produce some level of control of mimosa it is unlikely to achieve total control if used in isolation of other control methods. To date, eleven species have been released, including nine species of insects and two species of pathogenic fungi (Rea 1998). All have established in the field except for the most recently released seed-feeding insects, S*ibinia fastigiata* and *Chalcodermus serripes*, for which it is too early to confirm establishment. Although the agents released collectively damage vegetative and reproductive parts of the plant, mature leaves and roots are still largely undamaged, although they are heavily attacked by insects in the native range. Selection of further biological control agents is focusing on those that attack these plant parts.

Integrated control

Integrated control involves using a variety of control methods at a particular infestation site and can be successful if they use the cumulative benefits of individual control techniques, and decrease the probability of mimosa developing resistance to a particular control technique. A typical integrated control program would include appropriate survey and mapping, chemical control, mechanical control, and burning. Mechanical chaining and rolling of dead stems to compact the fuel may assist burning, or be a useful step before spraying with herbicides. The area should then be protected from grazing and fire for at least one year to allow the pasture to establish. Any regenerating mimosa plants should be spot treated and when livestock are introduced, grazing pressures should be closely monitored.

Possible control measures for Tram Chim

Although there is very little quantitative information on the distribution and spread of mimosa in Tram Chim and surrounding environments visual inspections and local knowledge can be used to identify areas that are currently heavily or lightly infested, or indeed, virtually free of mimosa. Given this situation a number of initial management strategies are outlined below.

Strategic control of mimosa

Survey

It is recommended that surveys to establish or confirm the extent of mimosa infestation in each sector of the Tram Chim National Park are undertaken. The survey information could include: date of recording; person recording; location; coordinates of the point or area

occupied by the infestation; description of location/habitat; estimated area of infestation; number/density estimate of plants; phenology of plants; control methods used; and results of previous control measures. The survey information should be stored in a formal record system, database and/or presented on a map.

Assessment

Undertake an assessment to identify priority areas for control activities. Prioritisation could be based on a number of factors, including: low level of current infestation; potential to become (further) infested; particular conservation value or use of the area; location within catchment; potential to spread to other sites; and usefulness as a demonstration site for training and public education.

Management measures

Recommended control methods (in brief) include: cutting and removal of flowers/seed pods; cutting and removal of stem material before flooding; hand-removal of seedlings (eg after draw-down or low level flooding); application of herbicides (eg foliar or basal bark application in association with above methods); and establishment of competitive plant species after physical removal of mimosa, in shallow water, or on areas exposed after draw-down.

Research

Research into specific aspects of the biology of the weed (eg timing of seeding and major growth periods) or specific control methods (eg stem cutting prior to flooding or the effectiveness of chemicals) may assist the development of the control program. This could be done in conjunction with an active control program, and should be coordinated to avoid confounding the results.

Public awareness and participation

Management of mimosa inside the Park can not be done effectively if it is isolated from the surrounding land and local communities. The Park is both a (potential) recipient and a source of propagules (eg seeds) for further infestation. Mimosa is also a direct threat to the livelihood of the local people as it can quickly spread along the banks of canals, streams and even paddies and prevent access by people. However, it is also a source of fuel-wood by local people. This resource could be used, given appropriate measures to ensure that it does not lead to further spread of infestations (eg by removing and burning the seed pods), to encourage local people to control mimosa near their houses etc and, under contract and supervision, in the Park.

Review and reassessment

Survey and reassessment of the program should be done on a regular basis. The reassessment will draw heavily on the records kept during the above described procedures. Where necessary the program should be adjusted, based on practical local experience and scientific evidence, and even stopped if proving ineffective (in terms of costs and results).

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Derivation of a site-specific water quality trigger value for uranium in Magela Creek[1]

RA van Dam[2]

Introduction

The revised Australian and New Zealand Water Quality Guidelines for Fresh and Marine Waters (WQGs) encourage the derivation of site-specific guideline trigger values (TVs) for toxicants (ANZECC & ARMCANZ 2000). Rather than supplying just a set of single numbers as guideline values, the WQGs provide a heirarchical decision framework from which default toxicant trigger values can be modified to suit local conditions. One option within the decision framework is to use local species toxicity data to derive a site-specific trigger value. This paper, adapted from van Dam (2000), describes an example of this approach for Magela Creek.

Toxicant trigger values

The process for deriving toxicant trigger values has changed from the previous WQGs, where a safety factor was applied to the lowest-observed-effect concentration (LOEC) of the most sensitive species tested (ANZECC 1992). The limitations of this approach have long been recognised (Warne 1998), with the revised WQGs adopting a modified statistical extrapolation method (Aldenberg & Slob 1993, Fox 1999, Shao 2000). The approach involves fitting the most appropriate distribution from the Burr Type III family of distributions to all no-observed-effect concentration (NOEC) data for a toxicant, to derive an estimated concentration that should protect at least *x*% of the species in the environment (Warne 1998, Shao 2000). Similar statistical distribution methods are used by the United States, The Netherlands, South Africa and Denmark, and are recommended for use by the OECD (ANZECC & ARMCANZ 2000). The percentage, *x*, can vary according to the level of protection afforded to the aquatic ecosystem of interest, with the current WQGs recommending a 95% level of protection for slightly to moderately disturbed ecosystems, and a 99% level of protection for ecosystems of high conservation/ecological value. By utilising all the toxicity data, a more confident estimate of a *safe* concentration is obtained. However, chronic NOEC data for at least 5 different species from at least 4 different taxonomic groups are required in order to derive a trigger value using the statistical extrapolation method. Where minimum data requirements are not met, the safety factor approach is used to derive the trigger value (ANZECC & ARMCANZ 2000).

At the time of publication of the WQGs, insufficient chronic toxicity data existed for uranium to enable the derivation of a trigger value based on the statistical extrapolation method. Subsequently, an interim, *low reliability* trigger value of 0.5 µg L^{-1} was derived using the less preferred safety factor approach (ANZECC & ARMCANZ 2000). This value was calculated

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[1] More detailed discussion of this research is provided in van Dam 2000, van Dam et al 2001 & 2002 (see 'Endnotes').

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by applying a safety factor of 20 to the lowest reported NOEC, being 10 μg L^{-1} for the freshwater cladoceran, *Moinodaphnia macleayi* (Hyne et al 1993). Given that the Magela Creek catchment is considered of high conservation/ecological value, a *low reliability* trigger value is considered inadequate, and site-specific assessment was considered essential. In addition, the interim trigger value is markedly lower than the Maximum Allowable Addition (MAA) under the current Ranger Authorisation for uranium in Magela Creek, of 3.8 μg L^{-1}, and would need to be accompanied by strong supporting evidence to be adopted.

Local species toxicity data

Since the mid 1980s, 21 freshwater species local to the Alligator Rivers Region (ARR) have been assessed for uranium toxicity (two cnidarian, one mussel, six crustacean, 10 fish and two plant species). However, until recently, there were insufficient chronic NOEC data to derive a site-specific trigger value based on local species toxicity data using the statistical extrapolation method. Many data were inappropriate because the studies did not assess chronic toxicity, or did not use natural Magela Creek water as the dilution water. Brief summaries of the available chronic toxicity data are presented below.

Chlorella sp.

In early 2001, the chronic toxicity of uranium to a local green alga, *Chlorella* sp. was assessed. The resultant NOEC and EC_{50} values (72-h cell division rate) were 129 and ~175 μg L^{-1}, respectively (Hogan et al in prep).

Moinodaphnia macleayi

Chronic uranium toxicity tests using the cladoceran, *M. macleayi*, in Magela Creek water were carried out in the early 1990s and again in the late 1990s, with the results being reasonably compatible. The NOEC values (3-brood reproduction) from tests in the early 1990s ranged from 14–22 μg L^{-1} (***eriss*** unpub data), compared with 8–29 μg L^{-1} in the late 1990s (Semaan et al 2001). The geometric mean of the NOEC values, being 18 μg L^{-1}, was taken to represent the NOEC of the species (as recommended by ANZECC & ARMCANZ 2000).

Hydra viridissima

Hyne et al (1993) assessed the chronic toxicity of uranium to green hydra, *H. viridissima*, in Magela Creek water. The NOEC and LOEC values (6-d population growth) were 150 and 200 μg L^{-1}, respectively.

Mogurnda mogurnda and *Melanotaenia splendida inornata*

Holdway (1992) assessed the toxicity of uranium to various life stages of the purple-spotted gudgeon, *M. mogurnda*, and the chequered rainbowfish, *M. splendida inornata*, over various exposure durations. For *M. mogurnda*, the lowest NOEC value (mortality) of 400 μg L^{-1} was obtained from a 7-day exposure/7-day post-exposure experiment using 1-day old larvae. For *M. splendida inornata*, the lowest NOEC value (mortality) of 810 μg L^{-1} was obtained following a 7-day exposure to 1-day old larvae.

Thus, based on historical and new toxicity data, NOEC values for five local species ranged from 18 to 810 μg L^{-1} (table 1).

Table 1 Summary of chronic toxicity of uranium to local species, using Magela Creek water as diluent

Species	Test endpoint	NOEC (µg L^{-1})	Reference
Chlorella sp.	Cell division rate	129	Hogan et al (in prep)
Moinodaphnia macleayi	Reproduction	18	***eriss*** unpubl, Semaan (1999)
Hydra viridissima	Population growth	150	Hyne et al (1992)
Mogurnda mogurnda	Mortality	400	Holdway (1992)
Melanotaenia splendida inornata	Mortality	810	Holdway (1992)

Deriving a site-specific trigger value for uranium

Using the toxicity data summarised in table 1, a site-specific trigger value was calculated by the software package, BurrliOZ, which was developed specifically for the WQGs. BurrliOZ uses a maximum likelihood method to determine which particular member of the Burr Type III statistical distribution best fits the toxicity data. It then calculates the concentration that will protect any specified percentage of species. The original methodology developed by Aldenberg and Slob (1993) used only the log-logistic distribution to model toxicity data, but Fox (1999) and Shao (2000) argued that the Burr Type III family of distributions provided a more flexible and defensible approach to deriving toxicant trigger values. In addition, the log-logistic distribution is actually a special case of the Burr Type III distribution, and thus, would be the distribution used if it was the one that best fit the data (Shao 2000).

Given that the Magela Creek catchment is considered of high conservation/ecological value, the WQGs recommend that a trigger value be calculated at the 99% protection level (ie 99% of species will be protected). Given that the value is calculated from NOEC data (not LOEC data), the trigger value is actually likely to offer more protection than prescribed. Using the local species NOECs from table 2, BurrliOZ calculated a 99% protection trigger value of 0.5 µg L^{-1}. This value was based on the Burr distribution, even though visual observation of the resultant plot (fig 1) indicated that the log-logistic and log-normal distributions appeared to be better approximations of the data. In theory, if the log-logistic distribution was a better fit then the trigger value should have been derived from this function, but in practice, this did not occur. This identified a significant error in the BurrliOZ software that the developers have since been working to rectify. It is thought that the method for determining the best fitting distribution is unreliable for small sample sizes.

Table 2 Observed versus predicted NOEC values from the Burr Type III and log-logistic distributions

	Predicted NOECs	
Observed NOECs	**Burr Type III**	**Log-logistic**
18	20	40
129	117	99
150	266	180
400	457	328
810	684	808
Correlation coefficient (r)	*0.970*	*0.989*
99% Trigger Value	*0.5 µg L^{-1}*	*5.8 µg L^{-1}*

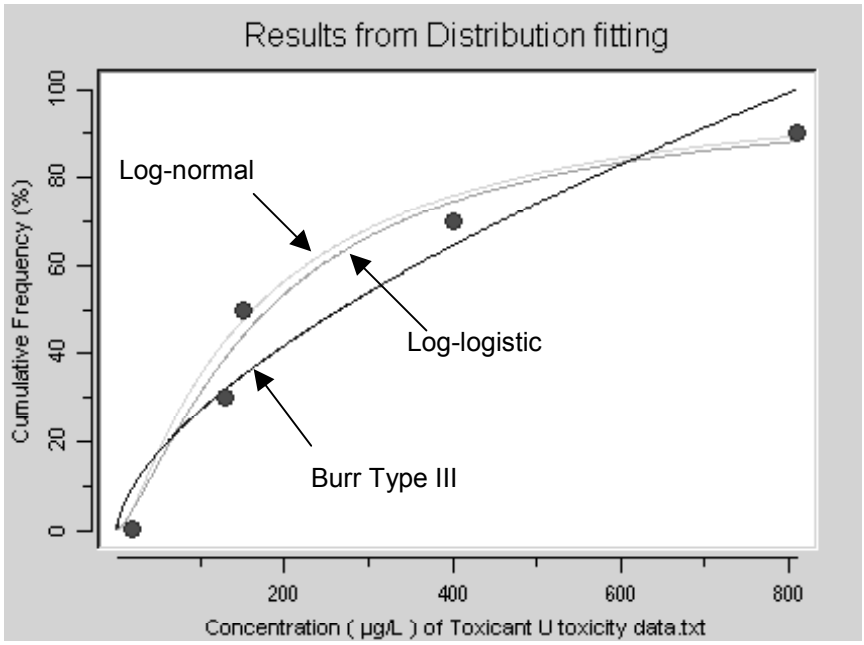

Figure 1 Graphical output of BurrliOz curve-fitting to uranium NOEC values for local species

In order to compare the chosen Burr Type III distribution and the log-logistic distribution, the latter was fitted to the toxicity data using *Minitab*, a statistical software package. The resultant plot is shown in figure 2. The 1st percentile, equivalent to the concentration to protect 99% of species was 6 µg L^{-1}, an order of magnitude higher than that derived using the Burr distribution. Correlation was carried out against the NOECs and the corresponding predicted values from both the Burr Type III and log-logistic distributions (table 2) in order to determine which curve best fitted the toxicity data. The correlation coefficients (r) for the Burr Type III and log-logistic distributions were 0.970 (P = 0.006) and 0.989 (P = 0.001), respectively, indicating that the log-logistic distribution was a better fit.

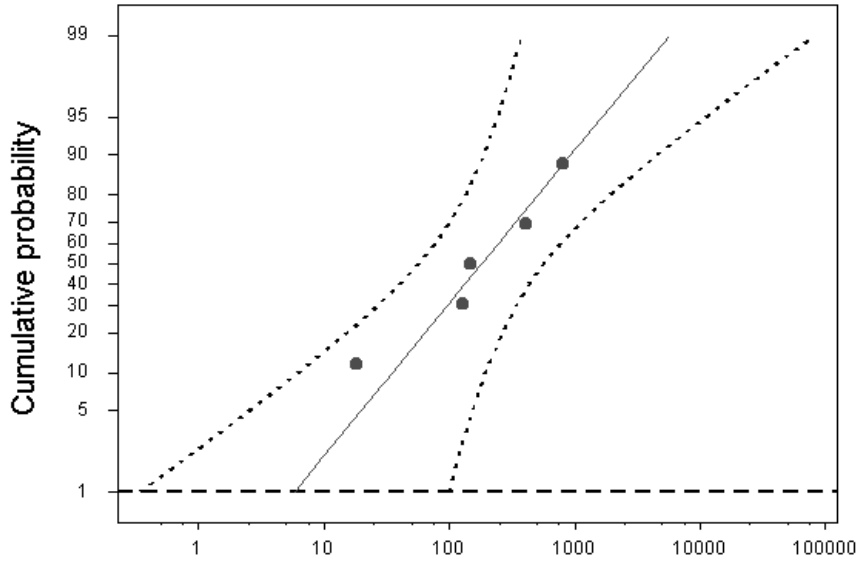

Figure 2 Log-logistic distribution fitted to uranium NOEC values for local species. Dotted lines represent 95% confidence limits.

Given that the log-logistic distribution provided a better fit to the toxicity data than the chosen Burr distribution, the trigger value of 6 μg L^{-1} was considered the more reliable estimate, and is recommended as the site-specific trigger value for uranium in Magela Creek.

The process undertaken here served to highlight the dangers in extrapolating to the tails of distributions that are based on few data points. The fact that the correlation coefficients for both distributions are highly significant, yet the resultant 99% protection level trigger values are an order of magnitude different, highlights the model-dependency of such values. Similarly, the calculation of toxicity point estimates below the 5–10% effect level has been criticised because the values are often model-dependent and possess large confidence intervals (Denton & Norberg-King 1996, Moore & Caux 1997). Increasing the number of data points will tend to decrease the error around the extrapolated value. Given this, there is a need, albeit not urgent, to obtain uranium toxicity data for a further three to five local aquatic species over the coming years. These will include an aquatic macrophyte, gastropod, mayfly and isopod species.

Conclusions

The revised Australian and New Zealand WQGs approach to deriving site-specific toxicant trigger values was applied to uranium in the Magela Creek system. Several flaws in the trigger value derivation approach and software were identified. Following a thorough analysis, a 99% protection level trigger value for uranium in Magela Creek was found to be 6 μg L^{-1}.

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The effect of silica on the toxicity of aluminium to a tropical freshwater fish[1]

C Camilleri, SJ Markich[2], BN Noller[3], CJ Turley, G Parker[4] & R van Dam[5]

1 Introduction

Gadjarrigamarndah (Gadji) Creek, in western Arnhem Land of northern Australia, has received acidic groundwater seepage, contaminated by spray irrigation of treated tailings water from the decommissioned Nabarlek uranium mine, for several years (van Dam et al 1999). A major consequence of groundwater acidification was the release of aluminium (Al) from soil minerals. Thus, since the spray irrigation period, Al has been measured in Gadji Creek water at concentrations of 40 to 540 µg L^{-1} (filterable fraction) at pH 4.2–7.2 (NTDME 2001), consistently exceeding the national guidelines (ie 1 µg L^{-1} at <pH 6.5; 55 µg L^{-1} at >pH 6.5; ANZECC & ARMCANZ 2000) for the protection of freshwater ecosystems.

Aluminium becomes more soluble and potentially more toxic to freshwater biota as pH decreases below 6.0 (Gensemer & Playle 1999). Although Gadji Creek water is generally acidic (pH 4.0–6.5) and contains elevated concentrations of Al, fish surveys from 1986 to 1995 have shown few differences in community structure and fish abundance, after an initial decline, compared to the pre-spray irrigation period (Pidgeon & Boyden 1995). Although Al levels were not directly compared, the results suggest that elevated Al concentrations in the surface waters of Gadji Creek have had no observable effects on the diversity and abundance of fish.

Factors known to reduce the toxicity of Al to freshwater fish include dissolved organic matter (eg humic substances), silica (Si) and fluoride (see review by Gensemer & Playle 1999). Birchall et al (1989) reported that in the presence of excess silica, as silicic acid (H_4SiO_4), the acute toxicity of Al to Atlantic salmon (*Salmo salar*) sac fry was eliminated at pH 5. In Gadji Creek, Si (as SiO_2) is typically 5 to 20 times the molar concentration of Al (NTDME 1996). Thus, the complexation of Al with Si may be reducing the toxicity of Al to fish in Gadji Creek.

The specific aims of this study were to:

i determine the toxicity of Gadji Creek water to a local native freshwater fish (ie purple spotted gudgeon, *M. mogurnda*) in the laboratory;

ii compare the toxicity data with the predicted speciation of Al in Gadji Creek water;

iii determine the toxicity of Al to *M. mogurnda* in the presence and absence of Si, to test the hypothesis that Al-silicate complexation reduces the toxicity of Al to *M. mogurnda*.

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[1] More detailed discussion of this research is provided in Camilleri et al 1999 & 2000 (see 'Endnotes').

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2 Materials and methods

2.1 Water sampling from Gadji and Cooper Creeks

Surface waters were collected from Gadji Creek (test water) and nearby Cooper Creek (control and diluent water) in August 1997 and September 1998. Upon arrival at the laboratory (<6 h after sampling) water for toxicity testing was filtered through a 10 µm paper filter (Whatman no. 91) and refrigerated (4°C) until required.

2.2 Preparation of test solutions using Gadji and Cooper Creek water

Test solutions were prepared using Cooper Creek water as diluent with the following dilutions: 0% (100% Cooper Creek water), 1%, 3.2%, 10%, 32% and 100% Gadji Creek water. The test solutions were stored in acid-cleaned 5 L polyethylene containers and refrigerated (4°C).

2.3 Preparation of laboratory test solutions

Reconstituted soft ASTM water (ASTM 1992) was prepared and used in the laboratory Al toxicity testing as control and diluent water.

The following Al concentrations were used for Al acute toxicity tests (Al Tests 1 and 2): 0, 250, 500, 750, 1000, 2000, 3000 and 4000 µg L^{-1}. In both tests, 4 mM 2-morpholinoethanesulphonic acid (MES; Good et al 1966) was used to maintain the pH of the water at 5.0 ± 0.2.

Two tests were carried out to determine the effect of silica on the toxicity of Al to *M. mogurnda* (Al Tests 3 and 4). Al concentrations were kept constant for each test. In Test 3, a constant Al concentration of 2000 µg L^{-1} was used with molar ratios of Si:Al (based on measured concentrations) being 0.5:1, 2.6:1, 5.0:1 and 9.2:1. The Al concentration in Test 4 was 1500 µg L^{-1} with molar ratios of Si:Al (based on measured concentrations) being 1:1, 4.7:1, 9.3:1 and 18.5:1. In Test 4, 4 mM MES was used to maintain the pH at 4.9 ± 0.2.

2.4 Toxicity testing procedures

Recently-hatched sac fry of the purple-spotted gudgeon, *M. mogurnda*, (<10 h old) were exposed to the above-mentioned dilutions of Gadji creek water, and concentrations of Al and Si, for 96 h. Sac fry were exposed to 30 mL of test water in acid-cleaned polycarbonate petri dishes. Three replicate dishes were used for each treatment (including the control), with each containing ten sac fry. The test dishes were maintained at 27 ± 1°C in a constant temperature incubator, with a photoperiod of 12 h light: 12 h dark. Test solutions were renewed every 24 h, following the recording of sac fry survival. The sac fry were not fed prior to, or during, the 96 h test. The test was considered valid if control survival exceeded 80% at the end of 96 h. Conductivity, pH and dissolved oxygen were measured daily on fresh (t_0) and 24 h old (t_{24}) test water.

2.5 Chemical analysis

The test waters were analysed for Na, K, Ca, Mg, Si, Al (total, filtered and labile), Fe, Mn (total and filtered), HCO3, Cl, NO_3, SO_4, total organic carbon (TOC) and dissolved organic carbon (DOC).

Measured concentrations of Al and Si were used to evaluate the concentration-response relationships.

2.6 Speciation modelling

HARPHRQ (Brown et al 1991), a thermodynamic geochemical speciation code, was used to calculate the speciation of Al in the test waters. The input parameters for HARPHRQ were based on physicochemical data (ie pH, redox potential and ion concentrations) measured in the test waters. Stability constants for Al species were derived primarily from Markich and Brown (1999). Additional stability constants for aluminium complexes with silica and MES (pH buffer) were calculated but are not shown here.

Aluminium complexation with dissolved organic carbon (humic substances) in Gadji Creek water was modelled using finite mixtures of simple organic acids, as described by Markich and Brown (1999). This approach has been shown to closely simulate metal binding to humic substances, the primary organic complexing agents, in natural waters.

2.7 Statistical analyses

Sigmoidal concentration-response relationships were fitted (where relevant) using a logistic regression model (Seefeldt et al 1995) for Tests 1 and 2. Using the model, the LC_{50} (ie the measured concentration of Al giving 50% survival over 96 h compared to the controls) and its 95% confidence interval (CI) were calculated. For Tests 3 and 4, one-way analysis of variance (ANOVA) and Dunnett's *post hoc* test were used to determine significant differences ($P \leq 0.05$) in sac fry survival from control treatments.

3 Results and discussion

3.1 Chemistry and toxicity of Gadji Creek water

Table 1 shows a comparison of water chemistry for Gadji Creek (August 1997 and September 1998) and Cooper Creek (reference water). For Gadji Creek water, the ionic composition varied between 1997 and 1998, with pH falling from 5.6 to 4.9 and dissolved (filtered) Al increasing from 33 to 137 µg L^{-1}. Given that the Australian guideline value for Al in freshwater at pH <6.5 is 1 µg L^{-1} (ARMCANZ & ANZECC 2000), measured values of Al in Gadji Creek exceeded the guideline on both sampling occasions. In comparison to Cooper Creek, Gadji Creek water generally has a lower pH and higher concentrations of ions (except bicarbonate) (table 1). The dissolved Al concentration in Cooper Creek in August 1997 (16 µg L^{-1}, pH 6.7) was below the freshwater guideline value of 55 µg L^{-1} at pH >6.5 (ARMCANZ & ANZECC 2000).

Gadji Creek water had no significant (P >0.05) effect on the survival of *M. mogurnda* sac fry in both August 1997 and September 1998, compared to control (Cooper Creek) water, with 100% survival in all treatments. These results are consistent with those of Hyne (1991) and Rippon and McBride (1994), who tested the toxicity of Gadji Creek water to *M. mogurnda* in 1991 and 1993, respectively. However, these studies did not relate their toxicity testing results to measured Al concentrations, nor other important water chemistry variables such as pH, Si or DOC. In accordance with the results of this study for *M. mogurnda*, van Dam et al (1999) found that 100% Gadji Creek water (August 1997) had no effect on the growth rate (96 h) of green hydra (*Hydra viridissima*), and only a small (-12%) effect on the reproduction (3 brood; 6 d) of the water flea, *Moinodaphnia macleayi*. In contrast, Rippon and McBride (1994) found that Gadji Creek water collected in April 1993 was highly toxic to *M. macleayi* and *H. viridissima*.

Table 1 Water chemistry of Gadji and Cooper Creeks

Parameter	Gadji Creek		Cooper Creek
	August 1997	September 1998	August 1997
pH	5.6	4.9	6.7
Conductivity (µS cm^{-1})	287	125	67
Na (mg L^{-1})	4.8	7.9	3.5
K (mg L^{-1})	1.3	0.3	0.1
Ca (mg L^{-1})	13	3.4	1.2
Mg (mg L^{-1})	20	7.8	5.6
Si (as SiO_2) (mg L^{-1})	17	13	6.9
SO_4 (mg L^{-1})	103	38	0.1
HCO_3 (mg L^{-1})	18	9.1	174
Cl (mg L^{-1})	5.5	8.6	4.9
NO_3 (mg L^{-1})	17	82	< 0.05
Total Al (µg L^{-1})	89	156	87
Dissolved Al (µg L^{-1})	33	137	16
Labile Al (µg L^{-1})	28	118	3.8
Total Mn (µg L^{-1})	67	34	10
Filtered Mn (µg L^{-1})	55	33	1.6
TOC (mg L^{-1})	3.5	4.1	3.6
DOC (mg L^{-1})	3.4	3.9	3.4

3.2 Predicted speciation of Al in Gadji Creek water

The predicted speciation of Al in Gadji Creek water (August 1997 & September 1998) is given in table 2. The results are based on the measured water chemistry variables given in table 1.

Table 2 Calculated percentage speciation of dissolved (filtered) Al in Gadji Creek water[a]

	% Al	
Al species	August 1997	September 1998
Inorganic Al species	7.2	39.5
Al3+ (%)	0.6	8.0
Al(OH)2+	1.7	5.4
Al(OH)2+	1.4	1.0
AlSO4	3.5	24.6
Organic Al species (Al-fulvate)	92.8	60.5

[a] Based on water chemistry given in table 1.

For both waters, the majority of Al (61–93%) was predicted to complex with humic substances (fulvic acid), where complexation was greatest in the water with higher pH (August 1997). Conversely, the formation of inorganic Al species (7–40%) was predicted to be greatest in the water with lower pH (September 1998). Of the inorganic Al species, $AlSO_4$ was dominant, given the elevated sulfate concentrations present in the water. These results are generally consistent with the those of other studies (Tipping et al 1991, Browne & Driscoll

1993) that have both measured and modelled Al in acidic waters with a similar chemical composition and organic carbon concentration.

Based on the results of the speciation modelling, bioavailable Al was estimated following the extended free ion activity model (Brown & Markich 2000), where bioavailable Al = $Al^{3+} \times 1 + Al(OH)^{2+} \times 0.67 + Al(OH)_2^{+} \times 0.33$. These monomeric species are more reactive, and hence toxic, at the cell membrane surface of aquatic organisms than polymeric forms and organically-bound Al (see review by Gensemer & Playle 1999). For Gadji Creek water collected in August 1997, bioavailable Al was estimated to be (0.7 µg L^{-1} (2.2% of the total dissolved Al concentration), which is below the national guideline value of 1 µg L^{-1} (ANZECC & ARMCANZ 2000). For Gadji Creek water collected in September 1998, bioavailable Al was estimated to be 16 µg L^{-1} (12% of the total dissolved Al concentration). Although the bioavailable concentration of Al was highest in water collected during September 1998, its toxicity to *M. mogurnda* was no different to water collected in August 1997. Therefore, it is possible that complexing of Al with other ligands, such as Si, SO_4^{2-} or humic substances, may have ameliorated the toxicity of Al to *M. mogurnda*.

3.3 Toxicity of Al to *M. mogurnda* in laboratory water

The concentration-response relationships for *M. mogurnda* sac fry exposed to Al at pH 5.0 ± 0.2 (Al Tests 1 and 2) are shown in figure 1. Values for the MDEC and LC_{50} are also given for each test.

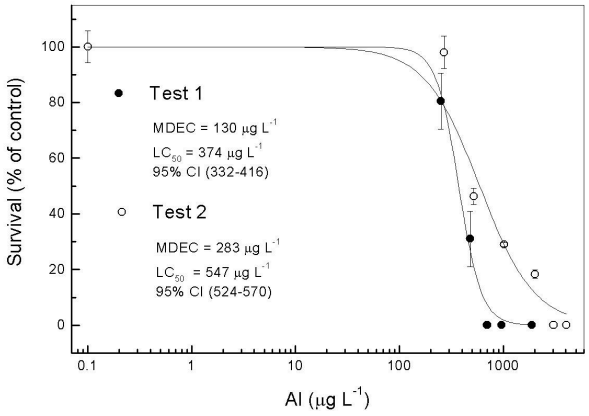

Figure 1 Concentration-response relationships for survival of *M. mogurnda* sac fry exposed to Al in laboratory water at pH 5.0. Data points represent the mean ± 95% confidence intervals. MDEC, minimum detectable effect concentration.

Despite the inherent variability in the endpoints between the tests, the LC_{50} values were comparable, albeit a little higher, to those reported for other fish species exposed to Al under comparable physico-chemical conditions (table 3).

The predicted speciation (% distribution) of Al in the laboratory test waters is given in figure 2. No organic complexing ligands were added to the test waters, apart from MES, which forms very weak metal complexes only. The formation of Al-MES was predicted to be negligible, comprising <1% of the measured Al concentration (not shown in figure 2). As shown for Gadji Creek water, $AlSO_4$ was the predominant inorganic Al species (60–64%) predicted to form. The concentration of SO_4 in the test water was relatively high (41 mg L^{-1}) due to the addition of $MgSO_4$ and $CaSO_4$ in the preparation of the reconstituted ASTM water. The use of non-sulfate salts of Mg and Ca (eg NO_3, which is non-complexing) for ASTM

water would probably increase the bioavailable fraction, and thus, the toxicity of Al to *M. mogurnda*. Increases in Al concentration resulted in only minor changes to the overall speciation of Al.

Table 3 Toxicity (LC_{50}) of Al to freshwater fish in soft acidic waters

Fish species	pH	Exposure (h)	LC50 (µg L-1)	Reference
Mogurnda mogurnda	5.0	96	374	This study
Mogurnda mogurnda	5.0	96	547	This study
Salmo salar	4.9	96	76	Roy & Campbell (1995)
Salmo salar	4.5	120	259	Roy & Campbell (1995)
Salmo salar	4.4	140	283	Roy & Campbell (1995)
Salmo salar	4.7	168	100	van Coillie et al (1983)
Salmo salar	5.3	168	170	van Coillie et al (1983)
Salmo salar	4.5	168	86	Wilkinson et al (1990)

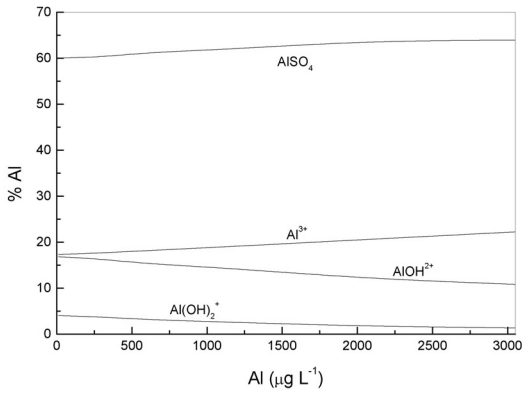

Figure 2 Predicted speciation (% distribution) of Al in laboratory water at pH 5.0

3.4 Effect of Si on the toxicity of Al to *M. mogurnda* in laboratory water

The effect of Si on the acute toxicity of Al to *M. mogurnda* sac fry in laboratory waters is shown in table 4.

Table 4 Acute toxicity (96 h) of Al to *M. mogurnda* sac fry in the presence of silica

Al Test 3			Al Test 4[a]		
Si : Al[b]	% Survival (95% CI)	pH	Si : Al[c]	% Survival (95% CI)	pH
0 : 0	93 (13)	5.1 ± 0.1	0 : 0	93 (7)	5.0 ± 0.1
0 : 1	0 (0)[d]	4.9 ± 0.1	0 : 1	0 (7)[d]	5.0 ± 0.1
0.5 : 1	67 (17)[d]	5.0 ± 0.1	1 : 1	40 (23)[d]	5.0 ± 0.1
2.6 : 1	100 (0)	5.3 ± 0.1	4.7 : 1	87 (7)	5.0 ± 0.1
5.0 : 1	100 (0)	5.5 ± 0.2	9.3 : 1	77 (7)	4.8 ± 0.1
9.2 : 1	93 (7)	5.9 ± 0.2	18.5 : 1	100 (0)	4.8 ± 0.1
			18.5 : 0	100 (0)	4.7 ± 0.2

[a] pH buffered with 4 mM MES; [b] 2000 µg L-1 Al; [c] 1500 µg L-1 Al; [d] indicates treatments that were significantly ($P \leq 0.05$) different to control treatments.

At fixed Al concentrations (ie Test 3, 2000 µg L^{-1} ; Test 4, 1500 µg L^{-1}) that were 3–4 fold the LC_{50} values, and in the absence of Si, zero survival of *M. mogurnda* sac fry was observed. As the ratio of Si:Al increased, the percentage survival of *M. mogurnda* sac fry increased, until a plateau was reached where there was no significant (P >0.05) difference from the controls (ie 0:0 Al:Si). Although the results from both tests are consistent, they are not directly comparable since the pH was tightly controlled (using MES) in Test 4 only. The pH of the water in Test 3 was observed to gradually increase (from 4.9 to 5.9) as the ratio of Si:Al increased. In Test 4, Si was added in the absence of Al (ie 18.5:0) to demonstrate that Si (27.7 mg L^{-1}) did not affect sac fry survival; indeed 100% sac fry survival was observed (table 4).

The results from Al Tests 3 and 4 clearly demonstrate that Si reduces the toxicity of Al to *M. mogurnda* at pH 5.0 (table 4). The results of this study are also consistent with those of Birchall et al (1989) and Exley et al (1997). Birchall et al (1989) showed that the acute (96 h) toxicity of Al to Atlantic salmon (*S. salar*) sac fry was eliminated at a Si:Al ratio of 13.5:1 at pH 5.0. Similarly, Exley et al (1997) reported that Si eliminated the acute (48 h) toxicity of Al to rainbow trout (*Oncorhynchus mykiss*) at pH 5.5. The latter authors provided evidence that at pH 5.5, the toxicity of Al is reduced by the formation of stable hydroxyaluminosilicates (HAS).

At pH 5.0 in the present study, the formation of stable HAS at the gill surface was not predicted using speciation modelling because the relevant reaction is kinetically, not thermodynamically, driven. However, the formation of AlH_3SiO_4 in solution was predicted to be minimal at pH 5.0 (ie 0.1% at 1:1 Si:Al to 2.3% at 18.4:1 Al:Si; figure 3), a finding confirmed experimentally by Pokrovski et al (1996) for natural waters. Thus, the speciation of Al in solution, and hence its bioavailability, was predicted to be constant as the ratio of Si:Al increased (figure 3).

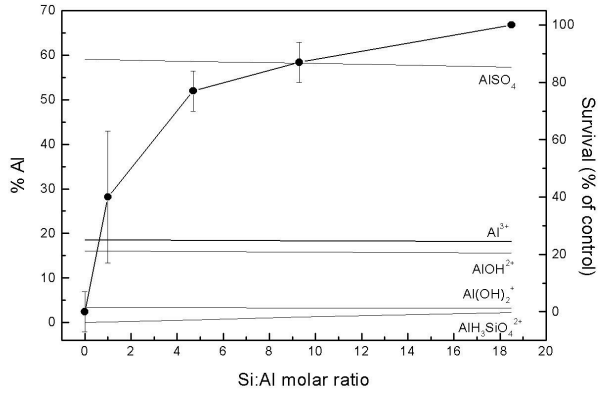

Figure 3 Predicted speciation (% distribution) of Al, together with the concentration-response relationship for survival of *M. mogurnda* sac fry, in laboratory water at pH 5.0 with an increasing Si:Al ratio (Test 4)

Therefore, there is no evidence to support the original hypothesis that Al-silicate complexes in solution reduce the toxicity of Al to *M. mogurnda*. According to the extended free ion activity model, the bioavailable Al in the test waters was also calculated to be constant (*ca.* 30%) as the Si:Al ratio increased, a result identical to the Al-only experiments. However, the acute toxicity of Al to *M. mogurnda* clearly decreased as the Si:Al ratio increased.

This apparent paradox may be interpreted as follows. Stable HAS may be forming at the gill surface and reducing Al toxicity by reducing the binding of free Al at the gill surface; although Exley et al (1997) found no evidence to support this at pH 5.0. This could be tested

directly by analysing for the presence of HAS by resin elution (Exley & Birchall 1993). Alternatively, excluding the formation of HAS, Si may be competing with free Al for binding sites at the gill surface. This could be tested by incorporating an Al radiotracer (^{26}Al) into the test waters and relating toxicity to metal uptake by the gills. A reduced uptake of ^{26}Al by the gills, together with a reduction in Al toxicity, would provide evidence to support the competition hypothesis.

4 Conclusions

Water from Gadji Creek, which has a low pH and contains elevated levels of Al and Si, was non-toxic to the sac fry of the purple spotted gudgeon, *M. mogurnda*, following acute exposure. It was hypothesised that the toxicity of Al to *M. mogurnda* was reduced by the formation of Al-silicate complexes. However, speciation modelling predicted that the majority of Al (85–96%) in Gadji Creek water was complexed with humic substances (ie fulvic acid) and sulfate, with less than 1% being complexed with silicate. Consequently, further experiments were undertaken to specifically assess Al toxicity and the effect of Si (in the absence of natural organic complexants) on Al toxicity. The addition of increasing amounts of Si to high Al concentrations (3–4 times the LC_{50}) clearly demonstrated that Si reduced, and even eliminated, the acute toxicity of Al to *M. mogurnda* at pH 5.0. However, speciation modelling again predicted very little Al (<3%) complexation with silicate, with the speciation and bioavailability of Al remaining constant as the Si:Al ratio increased. Therefore, there was no evidence to support the hypothesis that the formation of Al-silicate complexes reduces the acute toxicity of Al to *M. mogurnda* at pH 5.0. This, and an alternative hypothesis, that Si competes with Al for binding sites at the fish gill surface, are to be further investigated.

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Endnotes

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