supervising scientist report **166**

Environmental Research Institute of the Supervising Scientist Research Summary 1995–2000

edited by J Rovis-Hermann, KG Evans, AL Webb & RWJ Pidgeon

Environment Australia
Department of the Environment and Heritage

supervising scientist

J Rovis-Hermann – Environmental Research Institute of the Supervising Scientist, Locked Bag 2, Jabiru NT 0886, Australia.

KG Evans – Environmental Research Institute of the Supervising Scientist, GPO Box 461, Darwin NT 0801, Australia.

AL Webb – Office of the Supervising Scientist, GPO Box 461, Darwin NT 0801, Australia.

RWJ Pidgeon – Environmental Research Institute of the Supervising Scientist, GPO Box 461, Darwin NT 0801, Australia.

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Abbreviations/acronyms

ANSTO	Australian Nuclear Science and Technology Organisation
ANZECC	Australian and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
ARR	Alligator Rivers Region
AUSLIG	Australian Surveying and Land Information Group
EA	Environment Australia
ERA	Energy Resources of Australia Ltd
eriss	Environmental Research Institute of the Supervising Scientist
GIS	Geographic Information System
GPS	Global Positioning System
IPCC	International Panel on Climate Change
KNP	Kakadu National Park
NOEC	No-Observed-Effect-Concentration
NWQMS	National Water Quality Management Strategy
OECD	Organisation for Economic Co-operation and Development
oss	Office of the Supervising Scientist
SSD	Supervising Scientist Division
TV	trigger value
WQG	Water Quality Guidelines
WRD	waste rock dump

Acknowledgments

All papers appearing in this Research Summary have been peer reviewed. The Supervising Scientist would like to thank all reviewers for their input.

Overview

Max Finlayson

Director, Environmental Research Institute of the Supervising Scientist

Introduction

This report contains a summary of major research projects undertaken by staff from the Environmental Research Institute of the Supervising Scientist (***eriss***) over the 5 year period 1995–2000. It does not cover all research activities. These have been reported in various forms over this period, including external paper and reports (see 1995–2000 list in appendix 2), annual reports (appendix 3 contains all publications published by SSD), conferences and workshops, informal and local meetings, training courses and study tours for visiting scientits and environmental managers, and formal documentation submitted to the statutory Alligator Rivers Region Technical Committee.

The Research Institute is part of the Supervising Scientist Division (SSD) of Environment Australia. It was established to carry out independent research, on behalf of the Australian community, to establish the best methods available for the protection of people and ecosystems in the Alligator Rivers Region (ARR) both during and following mining in the region. Following the decision by the Australian Federal Government in 1993 to enact recommendations from the external review of the institute (Barrow 1994), we commenced a program of research on the ecology and conservation of wetlands. This recognised the skills available within the institute and the absence of a research unit principally addressing wetlands. This program has developed and is integrated within our research structure and is carried out largely with partners in the National Centre for Tropical Wetland Research which was formally established in November 1999. The Centre is a formal alliance between ***eriss***, James Cook University, Northern Territory University and the University of Western Australia.

In response to community concerns about environmental protection in the ARR we made consultation and communication tasks an integral component of our research activities. We also recognise that our research programs need to be developed in cooperation with stakeholders which include the communities potentially affected by mining activities in the ARR as well as the regulators, mining companies and wetland managers. This has seen consultation and communication tasks being formally included within our project planning and assessment. In particular we have taken steps to ensure that Aboriginal people in the region are included in these processes and where possible able to participate in research projects.

The research program

To fulfill the expectations from our research we undertook the following programs:

- Research on the impact of mining, particularly uranium mining, on people and ecosystems
- Research on the ecology and conservation of tropical wetlands
- Other environmental research as requested by Government

Research activities were divided into two branches — Environmental Impact of Mining and Wetland Ecology and Conservation. These were supported by a communciations program and corporate services. We also spend considerable time attending to formal governmental processes that both assist the implementation of our research programs and contribute to program, structural and personnel development within Environment Australia. Such activities include compliance with and promotion of occupational health and safety procedures, personal training and development, and staff assessment and performance. We also respond to requests for departmental briefs and information needs. The latter includes providing scientific guidance within various technical forums and national and international environmental agreements. In this respect we are not only keeping abreast with international practices but also providing leadership in environmental inventory, assessment and monitoring.

Workplans for the research program are reviewed each year and priorities assessed within general objectives. For the period 1995–2000 the research objectives were:

> ***Environmental Impact of Mining*** — provide advice, based on research and monitoring, to the Supervising Scientist and other stakeholders on standards, practices and procedures to protect the environment from the effects of uranium mining in the Alligator Rivers Region.
>
> ***Wetland Ecology and Conservation*** — provide advice, based on research abd minitoring, to key stakeholders on the ecology and conservation of tropical wetlands.

Based on these objectives and analyses of the required research effort the research programs were further divided into smaller teams to undertake specific projects and deliver the results to stakeholders. The ***Environmental Impact of*** **Mining** program comprised three units — environmental radioactivity; ecosystem protection; and erosion and hydrology. The ***Wetland Ecology and Conservation*** program comprised two units — wetland ecology and inventory; and wetland risk identification and assessment. Major projects undertaken by these units are described in this report. A listing of reports arising from these projects is also attached (see appendix 3).

References

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Introduction

Role of the Supervising Scientist

The position of the Supervising Scientist was established in 1978 under the *Environmental Protection (Alligator Rivers Region) Act 1978* to conduct research on the impact of uranium mining on the environment of the Alligator Rivers Region and to supervise the regulation of uranium mining in the region on behalf of the Commonwealth Government.

To assist the Supervising Scientist perform this role, the Environmental Research Institute of the Supervising Scientist (***eriss***) and the Office of the Supervising Scientist (***oss***) were established.

eriss conducts crucial research into the impact of uranium mining on the environment and the people of the Alligator Rivers Region, and on the protection and management of wetlands. ***oss*** carries out audit and policy functions. This Research Summary highlights some key areas of research undertake by ***eriss*** from 1995–2000. For more information about research undertaken during this time, please consult the Supervising Scientist Annual Report for each year (http://www.ea.gov.au/about/annual-report/index.html).

For information about the Alligator Rivers Region and the development of the uranium mining industry, please consult the list of publications produced by the Supervising Scientist Division (see appendix 2 & appendix 3) and the Supervising Scientist web site (http://www.ea.gov.au/ssd). A map of the Alligator Rivers Region is located on page 4.

Throughout the period 1995–2000, ***eriss*** comprised two research programs: 'Environmental Impact of Mining' and 'Wetland ecology and conservation'. Major topics included in these research programs are summarised in this report.

Environmental Impact of Mining

This program provides advice, based on research and monitoring, to the Supervising Scientist and other stakeholders, on standards, practices and procedures to protect the environment from the effects of mining, particularly uranium mining in the Alligator Rivers Region.

There were three main areas of research within this program:

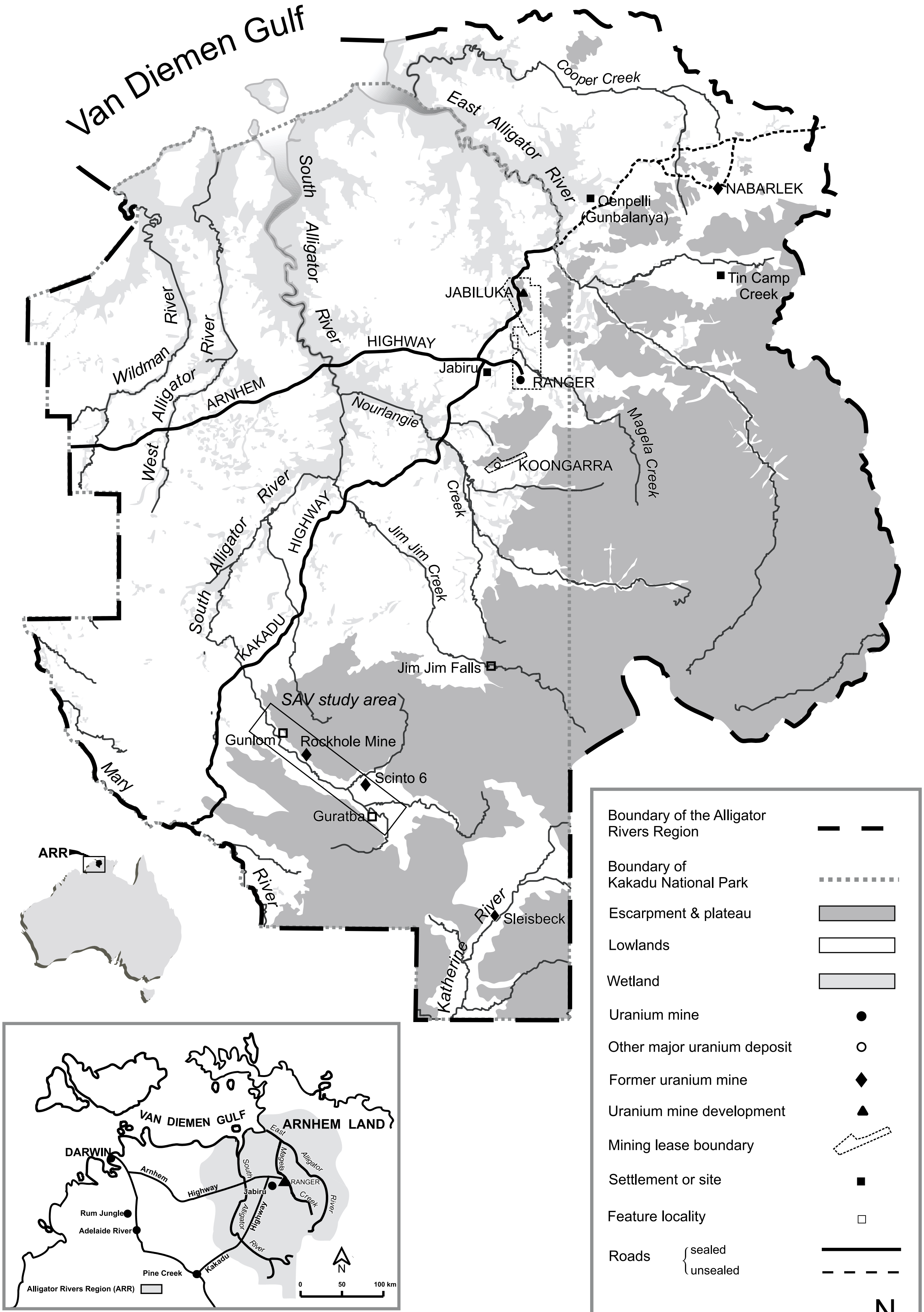
- Radiological Impacts of Mining
- Erosion and Hydrology
- Ecosystem Protection

Wetland Ecology and Conservation

This program provides advice, based on research and monitoring, to key stakeholders on the ecology and conservation of tropical wetlands.

There were two main areas of research within this program:

- Ecology and Inventory
- Risk Identification and Assessment



Supervising Scientist

Map of the Alligator Rivers Region

ENVIRONMENTAL IMPACT OF MINING

Radiological Impacts of Mining

Long-term study of groundwater dispersion of uranium at Ranger Mine

M Iles, P Martin, B Ryan & C leGras

Background

Energy Resources of Australia (ERA) has monitored uranium (U) concentrations in water from numerous bores located within the Ranger mine lease since the 1980s. It has been observed that the monitoring data have shown increasing U concentrations in filtered water from several of the bores, particularly those close to the north and north-eastern walls of the tailings dam (ie OB13A and OB16) (fig 1). This project aims to use the $^{234}U/^{238}U$ isotope ratio to indicate the source of uranium and to identify mechanisms to account for the concentration changes. This was possible as uranium-234 is more mobile than ^{238}U due to the physical recoil of the atom following the alpha decay of ^{238}U, and the displacement of the ^{234}U atom to a more chemically labile binding site. Consequently, the $^{234}U/^{238}U$ ratio is generally greater than unity in natural waters. In tailings dam water the ratio is close to unity following the chemical leaching process used in the milling of the ore (which extracts both isotopes with high efficiency).

Results and discussion

Locations of the bores sampled are shown in figure 1. Initially, based upon the results of ERA's monitoring data, only bores to the north and east of the tailings dam were targeted for study. However, more recently, the extension of waste rock dumps into this area has made the interpretation of results for these bores difficult. Consequently, recent work has concentrated on other bores on the Ranger site. These data will provide a baseline against which any possible future changes in U concentrations in these bores can be investigated.

Most of the U isotope ratio measurements have been made on samples collected since 1996. Figure 2 shows the $^{234}U/^{238}U$ isotope ratio results obtained for filtered water samples from a number of bores at Ranger plotted against the reciprocal of the U concentration. On such a plot a sloping trend line can indicate mixing of waters with different isotope ratios and different concentrations (Osmond & Cowart 1992). The resulting mix would lie on a line between the two end members. With high U concentrations and a $^{234}U/^{238}U$ ratio close to 1.0, tailings dam water would lie close to the bottom left corner of these plots (x~0, y~1). Therefore, a line with a *y*-intercept of 1 can indicate mixing with tailings water. Results are described below according to whether or not there appears to be some change in the source or behaviour of U in the bores.

No apparent source change

Since the beginning of the collection of bore monitoring data OB6A has consistently had elevated U concentrations compared to that measured at other Ranger observation bores. The $^{234}U/^{238}U$ ratios for OB6A have remained above 1.65, apart from May 1998 when a lower ratio (1.26 +/- 0.02) coincided with a higher than normal concentration.

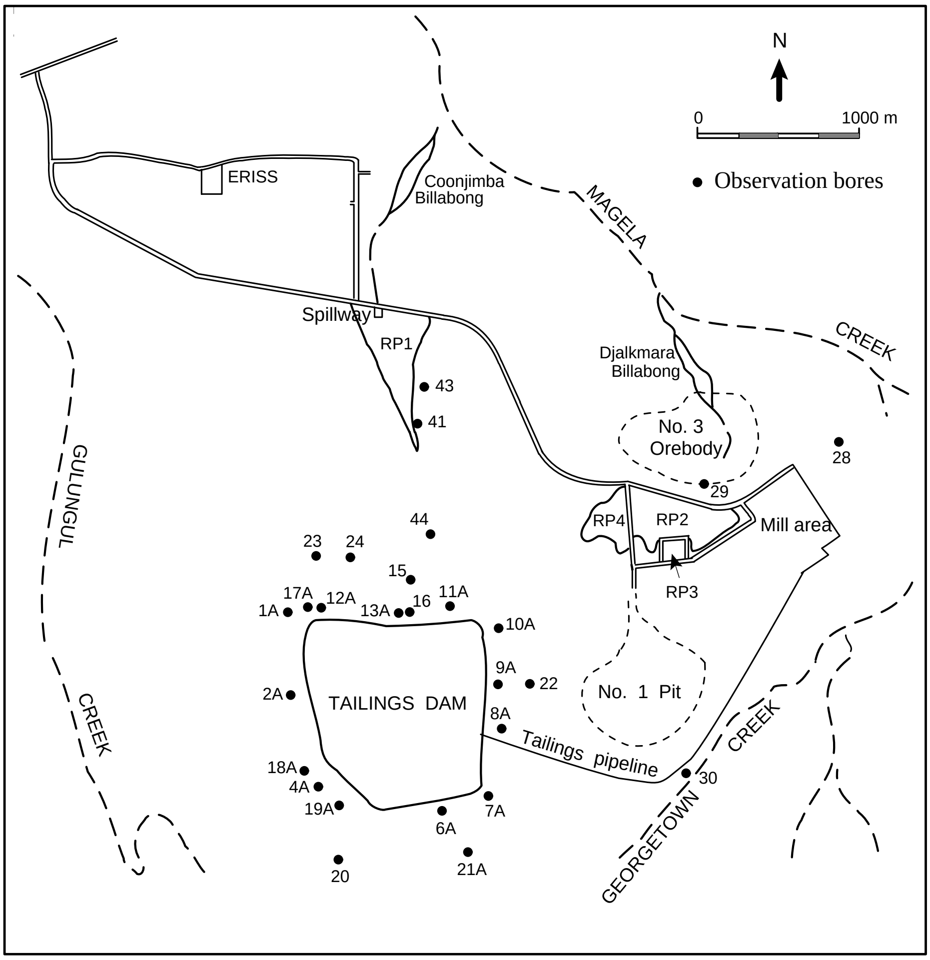


Figure 1 Location of observation bores at Ranger

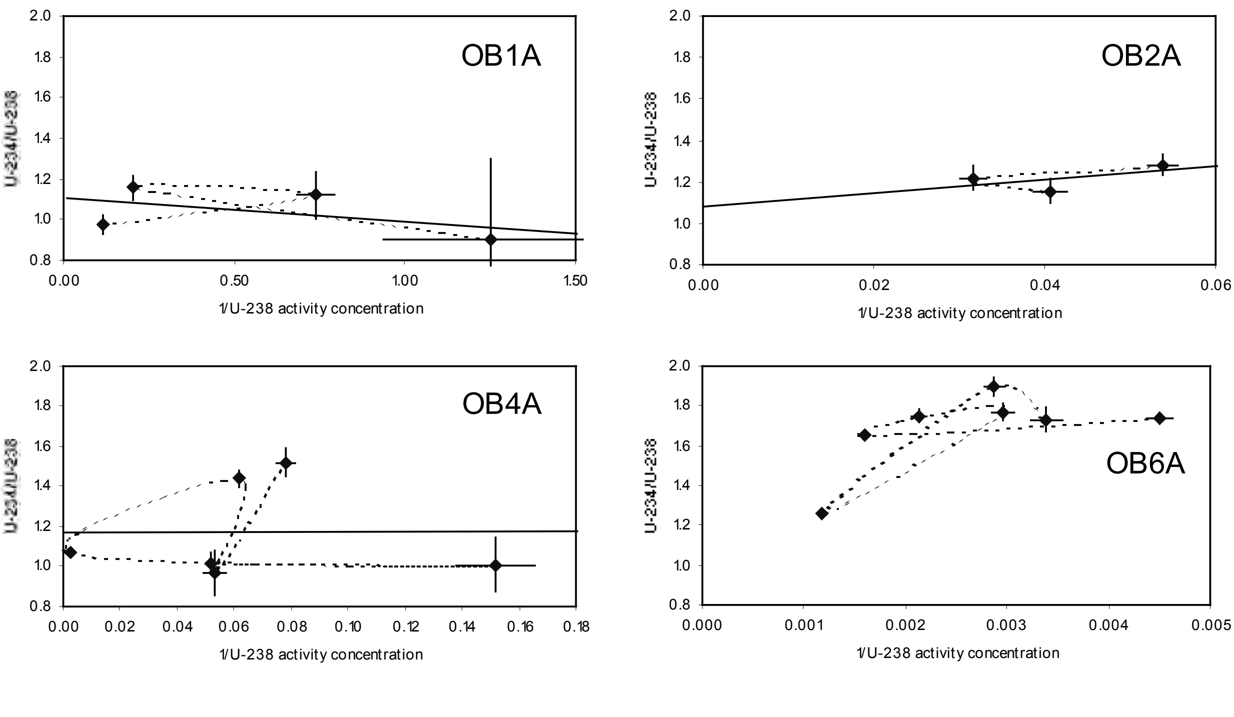
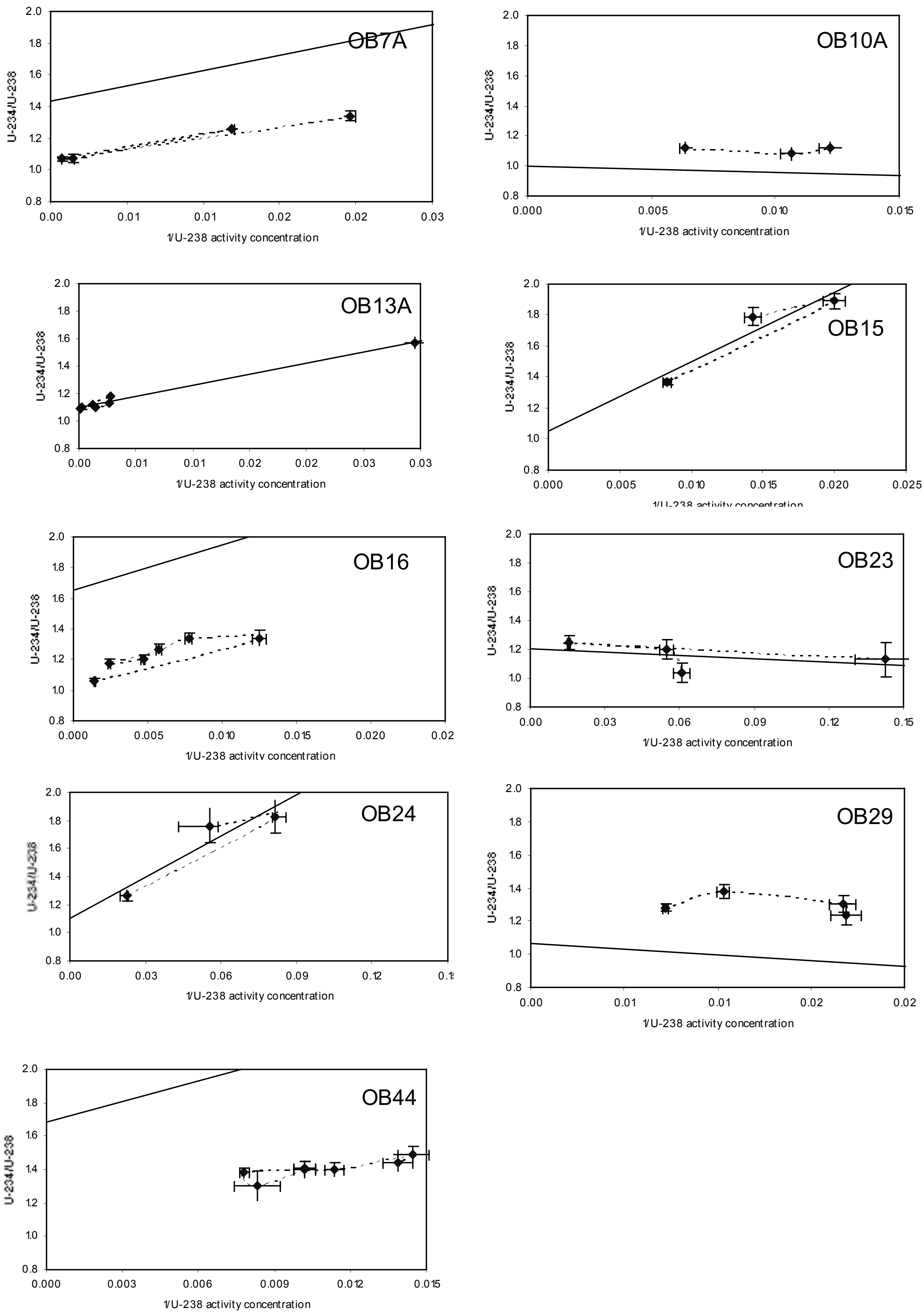


Figure 2 $^{234}U/^{238}U$ ratio vs. the reciprocal of the ^{238}U concentration obtained for filtered water from a number of Ranger observation bores. Error bars reflect counting statistics alone, and correspond to one standard deviation. The mixing line is shown in bold, the data sequence is shown by the dashed line.

Figure 2 continued



OB1A and OB4A show no correlation between U concentrations, which are variable in both bores and isotope ratios. The variability in U concentrations is most likely related to the high Fe concentrations for these bores, resulting in the formation of $Fe(OH)_3$ on sampling which removes the U from the filtrate fraction.

OB10A has relatively small un-correlated changes in both U concentration and isotope ratio values. OB23 and OB29 have fairly stable $^{234}U/^{238}U$ ratios un-correlated to U concentrations. OB2A and OB44 both appear to have fairly stable concentrations and isotope ratio values.

Possible Source Change

In OB13A the U concentration increase since the 1980s has been by a factor of about 1000; since about 1992 concentrations have often been of a similar order of magnitude to, or even greater than, the concentrations in tailings dam surface water. There is a strong correlation between this large concentration increase and a large decrease in the $^{234}U/^{238}U$ ratio.

No isotope ratio values are available for OB16 prior to the time of marked U concentration increase in 1990. Where isotope ratios are available their decrease shows some correlation with the concentration increases, but the measured value remained greater than or equal to 1.18.

OB7A oscillates between high and low U concentrations, which strongly correlate with the isotope ratio values, which decrease at times to 1.07.

Although there is only a small dataset for OB15 and OB24 (no. of samples = 3), there is a strong correlation between U concentration and $^{234}U/^{238}U$ ratios. The U concentration in OB15 has almost halved while for OB24 it has approximately doubled. The change in concentration in these bores, though significant, is relatively small compared to some other bores.

In the following, three proposed mechanisms for increased U concentrations are examined for consistency with the observed data.

Mechanism 1 — Transport of U from the tailings dam

Based upon time-series changes in U concentrations, and associated changes in isotope ratios, seepage could be affecting at least the following set of bores: OB7A, OB13A, OB15, OB16 and OB24. Figure 2 shows that the y-intercept values for each of these bores are greater than 1.0 implying that the U increases are not solely due to transport directly from the tailings dam.

The following observations also imply that direct seepage from the tailings dam is not dominating U behaviour in this system:

- The very high U concentration observed in November 1996 for OB13A was several times higher than the U concentrations in tailings dam water.
- The U concentration observed in some bores appeared to fluctuate during the year with U concentration values being higher late in the Dry season (November) than early in the Dry season (May). No such seasonal effect is seen in ERA's monitoring data for sulphate which is sourced primarily from the tailings (leGras et al 1993).

Mechanism 2 — Leaching of U from mine structure materials

Uranium concentration increases in OB13A and OB16, located close to the north wall of the tailings dam (fig 1), coincided with the raising of the tailings dam wall in 1990. Although not conclusive, the timing suggests that this was due to seepage from materials used in the dam wall construction, or altered groundwater flows due to the greater mass of the wall.

Waste rock dumps can contain unweathered, mineralised material which can be expected to degrade with exposure to rain and the atmosphere. As these waste rock dumps increase in height, the associated hydraulic head can be expected to increase and more material can be exposed to percolating rainwater which can dissolve U from the rocks. However, several observations do not support the rainwater percolation hypothesis, including: (1) the observed seasonal variations in U concentrations and ratios, (2) studies of the Ranger Land Application area have shown that U from the applied water is retained in the top few centimetres of soil (Akber & Marten 1992), and (3) U is comparatively immobile under the conditions expected in waste rock dumps, ie oxidising, near-neutral pH and relatively low conductivity.

Mechanism 3 — Mobilisation of U from aquifer rocks in the vicinity of the bore

The U isotopes should be equally labile when contained in carbonate minerals because these can be chemically leached with relative ease. Uranium mobilised from these minerals should therefore have a $^{234}U/^{238}U$ value near unity. Uranium-enriched carbonate facies are common in both Ranger ore and waste rock, and possibly also in aquifer material. If additional, highly labile ^{234}U was produced by recoil from a residual phase (such as silicate minerals), the isotope ratio may converge to near unity as carbonate leaching became the dominant source of U in borewater.

The available data do not support mobilisation of U from aquifer rocks as a dominant factor, ie the changes in U behaviour should be gradual and related primarily to changes in parameters such as the chemistry of the groundwater and the level of the water table. No simple relationship was observed between U concentrations, pH, conductivity and sulphate ion concentrations reported annually by ERA. However, the chemistry of U is very complex and a simple correspondence between these parameters and U concentrations cannot be expected. Also, displacement of either local or transported U from the aquifer walls by an ion exchange process would be preceded by displacement of a more exchangeable ion. No correlation was seen between measured sodium, potassium and U concentrations reported annually by ERA, although any ion-exchange effects would probably be masked by interactions with magnesium and calcium, which are present in much higher concentrations than U. It is possible that a relationship exists between U and ammonium ion concentrations but the latter has not been monitored.

Conclusions

At this stage, none of the three postulated mechanisms can be unequivocally given as the major influence on U behaviour in this system. However, the fact that y-intercepts for U ratio plots are generally greater than 1.0 implies that the primary source of the observed U increases is not the tailings dam. Interpretation of the results for bores which have shown increasing ^{238}U concentrations is made more difficult by the lack of an extensive $^{234}U/^{238}U$ ratio database for the period prior to the concentration increases. Consequently, work on this project is presently targeted at providing such a baseline dataset for other bores on the Ranger site.

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Development of a moving-grid dispersion model for radon and radon progeny

P Martin

Introduction

Atmospheric dispersion of Rn and its progeny and subsequent inhalation can be one of the dominant pathways for public dose from uranium mining. Most of this dose arises from exposure to the progeny rather than the Rn parent. Three important parameters for dose estimation are the Rn progeny potential alpha energy concentration (PAEC)[1], the equilibrium factor (E)[2] , and the unattached fraction (f_p)[3]. Measurements of Rn progeny for Jabiru East by Akber and Pfitzner (1994) show that the unattached fraction is unusually high with an annual average of about 14%, suggesting that effective dose estimates should be revised upwards (Akber et al 1994b). However, their reported measurements were not able to distinguish between mine-origin and non mine-origin PAEC.

In this section, the development and validation of a model for the atmospheric dispersion of Rn and Rn progeny following exhalation of Rn from defined sources will be described. The majority of reported studies on behaviour of Rn progeny in the air relate to the indoor environment, while many outdoor studies deal with temperate and urban environments. This makes the choice of values for many of the parameters of the model problematic for a tropical, rural location such as Jabiru. Since the main purpose of the present work is to estimate whether or not the unattached fraction could make a significant difference to dose estimates in the ARR, a set of values for input parameters for the model has been derived from the literature which can be expected to be approximately correct. A sensitivity analysis has then been carried out to determine which of these parameters gives rise to the greatest uncertainties in model predictions (and hence which parameters would be the most important to address in future research on Rn and Rn progeny dispersion).

Model description

Description of the moving-grid dispersion model

The dispersion model uses a variation of the grid-cell method, referred to here as a *moving-grid* model. This uses a 2-dimensional grid of cells, and calculates the transfer of Rn and progeny successively from one grid to the next downwind of the source. A computer program, *RAPAD* (Radon And Progeny Atmospheric Dispersion model), has been written in the C programming language to carry out the model calculations (Martin 2000).

---

1 PAEC is essentially a measure of the total amount of alpha energy which would be liberated upon the decay of all of the short-lived Rn progeny present in a unit volume of air.

2 E is essentially the ratio of the actual PAEC to that which would be present if the progeny were in secular equilibrium with the Rn parent.

3 The unattached fraction, f_p, referred to here corresponds to the ratio of unattached PAEC to the total PAEC, where 'unattached' refers to PAEC associated with an aerosol with diameter less than approximately 4 nm.

Calculation of dispersion

The x dimension (ie in the current wind direction) of each cell was 100 m, the y (horizontal, perpendicular to x) dimension was 100 m and the z (vertical) dimension was 10 m (ie the approximate height of the tree canopy). For calculation of the proportions to pass to the cells of the next grid, a Gaussian distribution in the y and z directions has been assumed. Increase in wind speed with height has been calculated from the power law:

$$U(z) = U(z_0)\left(\frac{z}{z_0}\right)^n$$

where $U(z)$ is the wind speed at height z (m), $U(z_0)$ is the wind speed observed at reference height z_0, and n is an empirical constant taken here to be 0.17 (Boeker & van Grondelle 1995). Predictions of mixing layer height $H_m(T)$ were based on the formulation suggested by Petersen et al (1992). Values ranged from 40 m to 1600 m, depending on time of day and wind speed.

Calculation of ingrowth/decay/attachment/removal

Figure 1 shows the model used for calculation of ingrowth, decay, attachment and removal of Rn and Rn progeny. λ_1 ... λ_5 represent the radioactive decay constants for ^{222}Rn … ^{214}Po, while λ_A is the rate constant for attachment of Rn progeny. λ_{RU} and λ_{RA} are the rate constants for removal of Rn progeny by rain for unattached and attached progeny, respectively. λ_{DU} and λ_{DA} are the corresponding constants for dry deposition.

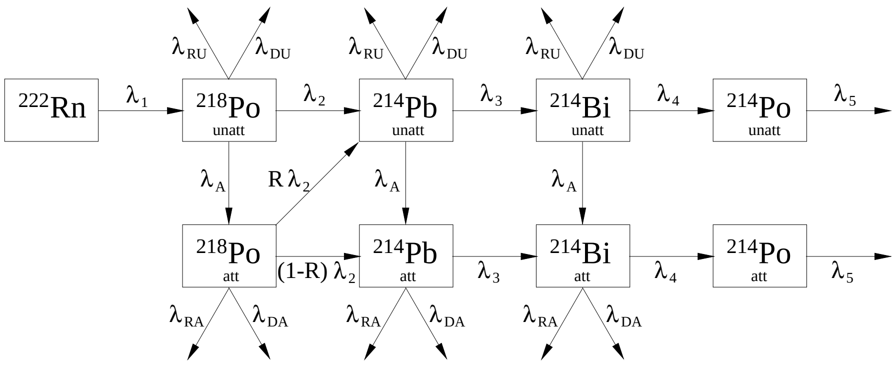

Figure 1 Ingrowth/decay and attachment/detachment model for Rn and Rn progeny

R is the probability of attached ^{218}Po atoms becoming unattached following α decay due to recoil and has been taken as 0.8 (Porstendörfer 1994). The attachment rate constant (λ_A) is calculated from the product of β, the attachment coefficient of Rn progeny to an aerosol particle [cm^3 s^{-1}], and CN, the condensation nuclei number concentration [cm^{-3}]. β has been estimated from measurements for the open atmosphere reported by Porstendörfer and Mercer (1978).

Rate constants for removal of attached and unattached progeny by dry deposition were estimated using values published by Butterweck (1991). Rainfall scavenging rate coefficients have been estimated from the semi-empirical derivation of Seinfeld (1986).

Input data

A 1-year meteorological dataset collected at Jabiru East between February 1989 and February 1990 was used. This dataset has hourly records for wind speed, direction and σ_θ (wind direction standard deviation) at 17 m height and for rainfall at ground level. Condensation nuclei concentrations were obtained from the study of Akber and Pfitzner (1994).

Akber et al (1993) reviewed the available data for Rn exhalation fluxes from the Ranger operation, and their recommended values have been used. These were: 300, 3760, 175, 1730 and 1090 kBq s^{-1} for pit 1, the ore stockpile, waste rock dump, mill plant and tailings dam, respectively.

Results and discussion

Table 1 shows a comparison of model predictions with experimental results (Akber et al 1994a) for the yearly average Ranger mine-derived signal for ^{222}Rn and Rn progeny PAEC at 15 m height for Jabiru East and Jabiru Town. Unfortunately, experimental data for the mine-derived unattached fraction are not available for comparison with the predictions of the model.

Overall, the agreement between model and experimental results is reasonable (about a factor of two), and it would be advisable to obtain improved input parameters and to carry out testing at other sites before further refinement of the dispersion modelling approach is attempted.

Table 1 Comparison of model predictions and experimental results (Akber et al 1993, 1994a) for the yearly average mine-derived signal at 15 m height for Jabiru East and Jabiru Town, March 1989 to February 1990

	^{222}Rn (Bq m^{-3})		Progeny PAEC (nJ m^{-3})	
	Model 10–20 m	Expt 15 m	Model 10–20 m	Expt 15 m
Jabiru East	4.3	7	5.6	9.8
Jabiru Town	2.5	2	4.2	3.1

Table 2 shows the yearly average results from the model for the grid height 0–10 m at a number of human occupation sites (arranged in order of increasing distance from Ranger). As expected, ^{222}Rn concentrations generally decrease with distance from Ranger. The low predicted concentrations for Magela 009 and Mudginberri are due to the fact that the predominant wind direction is from the east/south-east, whereas Magela 009 and Mudginberri lie towards the north of Ranger. The predicted equilibrium factors increase with distance from Ranger, while the unattached fractions decrease. This is due to the increasing time available for ingrowth and attachment of Rn progeny with greater travel distance.

Table 2 Summary of model predictions for the yearly average mine-derived signal at 0–10 m height for Jabiru East, Magela 009, Jabiru Town and Mudginberri, Feb 1989 to Feb 1990

Location	^{222}Rn (Bq m^{-3})	Progeny PAEC (nJ m^{-3})	Equilibrium Factor *E*	Unattached Fraction f_p
Jabiru East	6.0	2.9	0.09	0.43
Magela 009	1.2	1.0	0.16	0.27
Jabiru Town	3.1	2.6	0.16	0.29
Mudginberri	0.8	0.8	0.18	0.24

The results from the model runs imply that f_p will be high for the Ranger mine-derived fraction of Rn progeny at the relevant living areas. This finding supports the conclusions of Akber and Pfitzner (1994), based on experimental data, that (total) f_p values for the open air in the ARR are unusually high. The reason for this is that the area is remote from major industrial activity (apart from the Ranger mine itself), and so condensation nuclei concentrations are very low.

Sensitivity to model parameters

Table 3 shows the results of a sensitivity analysis of the model to twelve of the main model parameters (the Wet season month of December 1989 was used in this analysis). The table shows the change in predicted values for mine-derived ^{222}Rn concentration, equilibrium factor and unattached fraction at Jabiru Town (0–10 m height) when each parameter is independently increased by 10% from its base value.

Table 3 Variation (%) in predictions for Jabiru Town with an increase of 10% in various model parameters, calculated for the meteorological dataset for December 1989. Calculation of the base values for σ_y, σ_z, $H_m(T)$ and Q is complex and is detailed in Martin (2000).

Parameter	Base value	^{222}Rn	Equilibrium factor E	Unattached fraction f_p
Attachment rate constant β ($cm^3\ s^{-1}$)	6.0×10^{-7}	0	3.7	-5.3
Unatt. removal rate by rain λ_{RU} (s^{-1}/(mm hr^{-1}))	1.4×10^{-6}	0	-1×10^{-4}	-4×10^{-5}
Att. removal rate by rain λ_{RA} (s^{-1}/(mm hr^{-1}))	8.0×10^{-9}	0	-7×10^{-7}	7×10^{-7}
Unatt. deposition λ_{DU} (s^{-1})	2.2×10^{-2}	0	-1.9	-2.2
Att. deposition λ_{DA} (s^{-1})	3.3×10^{-4}	0	-1.7	1.7
α-recoil detachment R	0.8	0	-1.1	1.5
Wind speed exponent n	0.17	4.5	1.5	-1.6
Horizontal dispersion σ_y		0.41	0.27	-0.16
Vertical dispersion σ_z		-2.9	3.2	0.66
Mixing layer height $H_m(T)$		-1.1	-0.88	0.59
Rn exhalation fluxes Q		10.0	0	0

Predictions for ^{222}Rn concentrations were found to be most sensitive to changes in the source Rn exhalation fluxes, the wind speed exponent (n) and vertical dispersion (σ_z and $H_m(T)$). Equilibrium factor and unattached fraction predictions were most sensitive to changes in the attachment rate constant, deposition rate to the ground, and vertical dispersion (σ_z and $H_m(T)$). As a result of this analysis, these factors will be targeted in future projects on atmospheric dispersion of radon and progeny in the region.

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Radon exhalation rate from the rehabilitated Nabarlek surface

P Martin, S Tims & J Storm[1]

Introduction

One of the major data requirements for atmospheric dispersion models is detailed knowledge of the source term. In the case of radon dispersion in the environment, this is the radon exhalation rate from the ground surface. Ideally, variation both geographically and with time (ie season and time of day) should be known.

This paper describes research undertaken to obtain improved estimates of radon exhalation rates in the Alligator Rivers Region, particularly from the rehabilitated Nabarlek site. Once this work is completed, the results of dispersion models will be compared with measurements being made of radon concentrations in air at and near the site. Three separate sub-projects will be discussed here, relating to geographic variability over the Nabarlek site, and seasonal and diurnal variability.

In this paper, 'radon' refers specifically to the isotope ^{222}Rn, while 'thoron' refers to ^{220}Rn. ^{222}Rn is a member of the uranium decay series and has a halflife of 3.8 days, while ^{220}Rn is a member of the thorium decay series and has a halflife of 56 seconds.

Geographic variability study at Nabarlek

Figure 1 shows the layout of the Nabarlek minesite during operations. No separate tailings dam was required since tailings could be placed directly in the mined-out pit. Following completion of milling operations in 1988, the tailings were covered by geotextile followed by a graded rock and leached sand layer of 1 to 3 metres. With final decommissioning of the mine in 1995, remaining contaminated material and unsaleable plant equipment were placed in the pit and covered with another layer of waste rock. Most of the other mine areas (with the exception of the topsoil stockpile and plant areas) were left covered with run-of-mine waste rock.

The work described here was undertaken to give information on the radon exhalation rate from this rehabilitated area, including information on the geographic variability over the site. The measurements were made in the late Dry season of 1999 (August/September) since the soil moisture content, and hence also the radon exhalation rate, should be reasonably stable at this time of year.

Measurements were carried out using the charcoal cup technique. Cups containing activated charcoal were left in place over a 3-day period before removal and counting on a γ-ray spectrometer. This method gives a measure of the exhalation rate of radon but not of thoron.

[1] John Storm, Department of Physics, University of Adelaide, Adelaide, SA 5005, Australia.

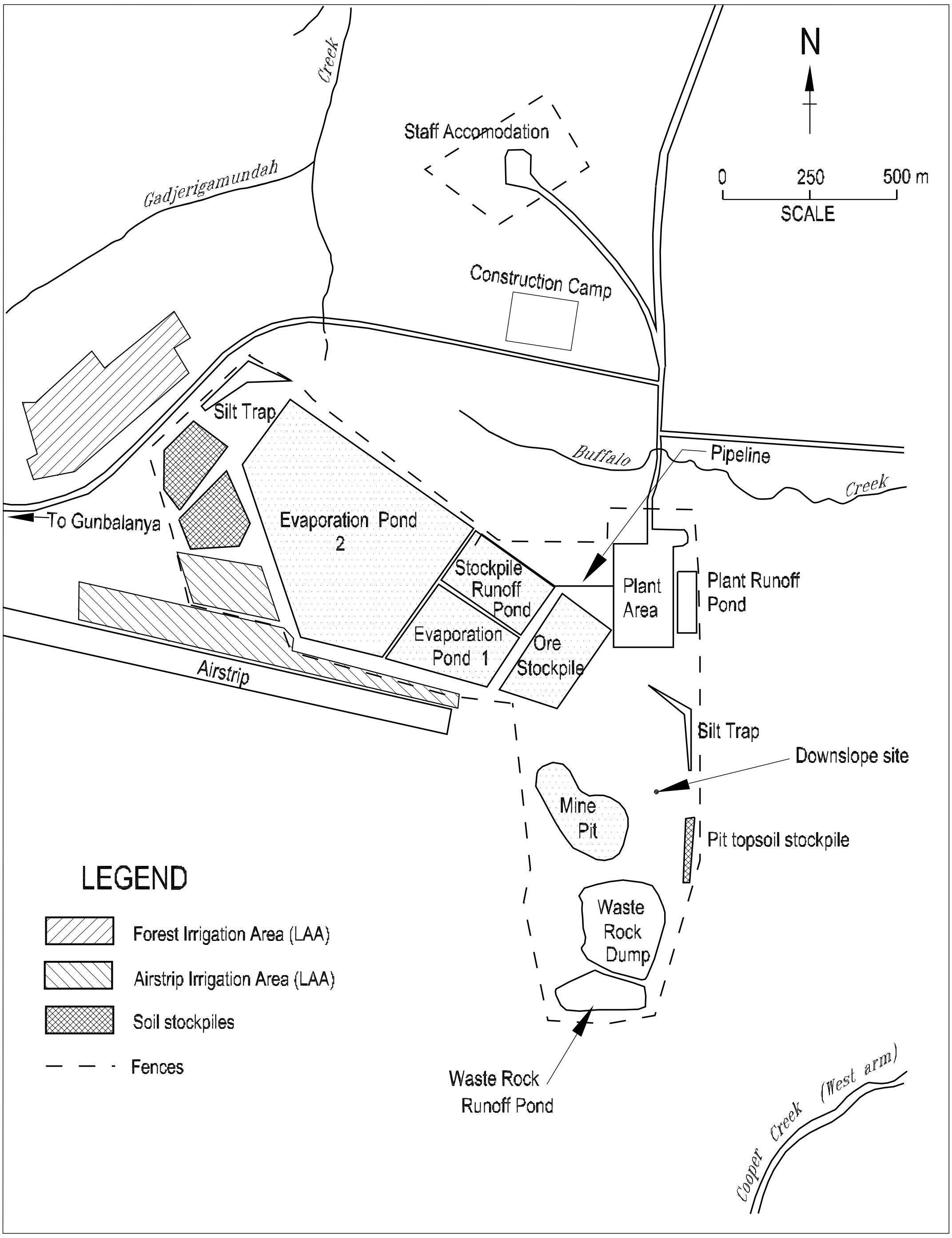

Figure 1 Map of the Nabarlek site showing locations of major areas during operations, and the 'downslope' radon exhalation study site

Table 1 shows the results for radon exhalation rate for measurements made over the former locations of various structures on the Nabarlek fenced area. Also given is the total radon flux (in kBq s^{-1}) over each of these areas. Of these, the former pit gives rise to the greatest radon flux despite its small area (5 ha), due to its relatively high exhalation rate. It is unlikely that this higher exhalation rate is due primarily to radon sourced from the tailings, since these are approximately 13 metres below the final ground surface, and are below the ground water level (Waggitt & Woods 1998). A more likely reason is the substantially greater depth of waste rock over the pit, compared with the other areas.

Table 1 Original area, and measured post-rehabilitation radon exhalation rates and radon flux, for major areas of the Nabarlek site

Location	Area (ha)	^{222}Rn exhalation rate (mBq m^{-2} s^{-1})			^{222}Rn flux (kBq s^{-1})
		Mean	Standard deviation	*n*	
Pit	5	971	739	42	49
Plant Runoff Pond	1.1	278	203	24	3
Ore Stockpile	6	77	59	21	5
Stockpile Runoff Pond	3	137	120	18	4
Evaporation Pond 1	5	169	86	12	8
Evaporation Pond 2	25	103	102	83	26
Topsoil Stockpiles	7	31	28	17	2

Seasonal variability study

Measurements of radon in air in the Alligator Rivers Region show that concentrations are lower in the Wet season than in the Dry season by about a factor of two to three (Akber & Pfitzner 1994). It was expected that at least part of this variability is due to variation in radon exhalation rates due to changes in soil moisture content. This sub-project aims to obtain quantitative data on this variability by measuring radon exhalation rates from a limited number of sites over the course of a yearly cycle.

The main part of this work is planned to be carried out in 2002 and 2003, however, some preliminary data have been collected from one site to give an indication of the variability which could be expected. The site is a forest area east of Baralil Creek near Jabiru Town. Measurements have been made at approximately monthly intervals between September 2000 and February 2001. The instrument used gives the exhalation rates (over a measurement period of 30 minutes) of both radon and thoron.

Figure 2 shows the results obtained for radon and thoron exhalation rates, as well as soil moisture content measured from a 5 cm depth sample taken close to the measurement site (ensuring that the exhalation site itself is not disturbed). The strong influence of soil moisture on the radon exhalation rate is apparent, with the rate falling from about 35–45 mBq/m^2/s in September/October, through about 30 mBq/m^2/s in November/December, to 0 ± 1 mBq/m^2/s for a 15% moisture content on the February measurement.

The thoron exhalation rate also fell markedly in February. Most of the thoron can be expected to originate from the top 1 to 2 centimetres owing to its very short halflife; the change in February was therefore consistent with a hypothesis that the high moisture content was very effective at retaining radon and thoron within even the top layer of the soil.

These results demonstrate that seasonal variability in radon and thoron exhalation rates can be dramatic, giving strong evidence that this is an important factor in variation of concentrations in air. However, exhalation rate variability can be expected to vary from location to location, depending on such factors as the local drainage pattern and soil porosity. These will be particularly important when considering rates from minesite areas such as waste rock dumps.

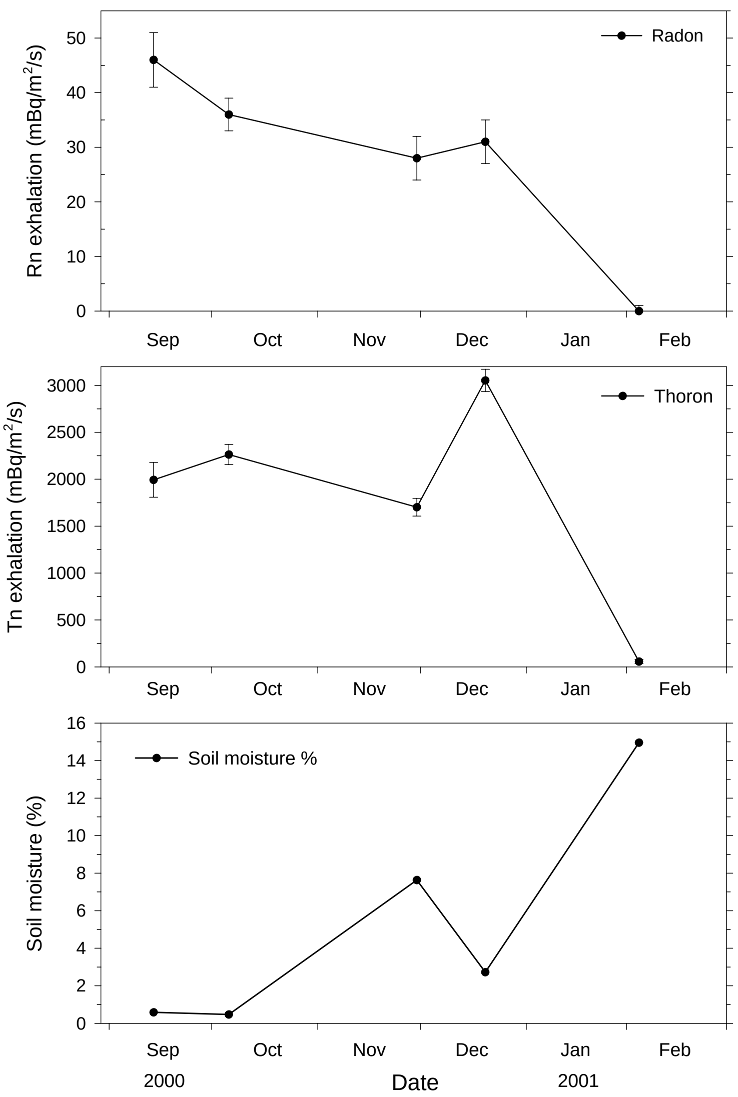

Figure 2 Seasonal variability in radon and thoron exhalation rate, and surface soil moisture, at the Baralil Creek forest site. Error bars reflect counting statistics alone, and correspond to one standard deviation.

Diurnal variability study

Few data are available in the literature on diurnal variability in radon exhalation rates. Todd et al (1998) measured radon and thoron exhalation rates over the course of a 24-hour period at a site at Jabiru East. Their results were equivocal, mainly due to the relatively low radon exhalation rate from this site (giving rise to large counting statistical errors), but indicated that if a diurnal cycle occurred for radon it was no greater than about 20% of the mean exhalation rate at this site.

A separate series of measurements were made between the 16th and 17th June 1998 at the 'downslope site' at Nabarlek (see fig 1). This site was chosen for study due to its relatively high radon exhalation rate. The instrument used was the same one used in the above seasonal variability measurements. Figure 3 shows that there was no apparent cycling for either radon or thoron exhalation rate over the daily cycle. A two-tailed t-test ($\alpha = 0.05$) did not show a significant difference between daytime and nighttime exhalation rates for either radon or thoron.

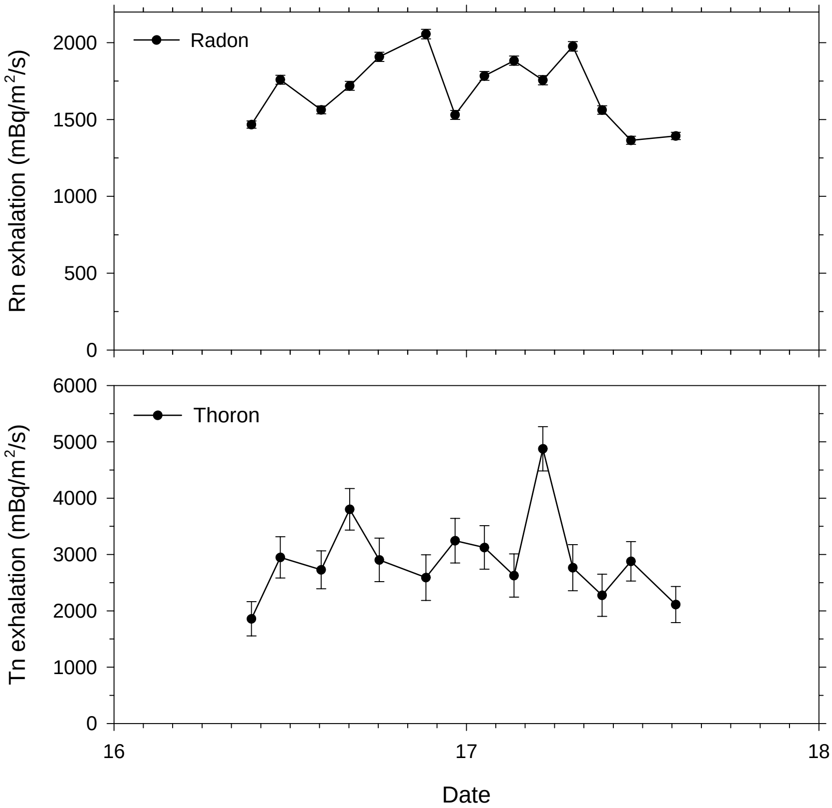

Figure 3 Test of diurnal variability in radon and thoron exhalation rates at the Nabarlek 'downslope site'. Error bars reflect counting statistics alone, and correspond to one standard deviation.

Conclusions

Data have been obtained for the radon exhalation rate from major areas of the rehabilitated Nabarlek minesite for the August/September period. The majority of the radon flux from the measured areas is sourced from above the former pit.

Preliminary data show that the difference in radon exhalation rate between Dry and Wet seasons can be large. However, this variability is expected to depend upon site-specific factors and so further, detailed studies are planned. The available data on diurnal variability indicate that it is not large, at least during the Dry season.

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Identification of traditional Aboriginal foods for radiological assessment

B Ryan & P Martin

Introduction

Previous studies conducted by ***eriss*** of radionuclides in Aboriginal bushfoods have focussed on bioaccumulation by aquatic animal and plant species (Pettersson et al 1993, Martin et al 1998). This was due to the importance of the aquatic transport pathway, particularly during the operational phase of uranium mining operations. Despite the work which has been carried out to date, there remain several significant areas of uncertainty, including the following:

- Knowledge of the bushfood consumption by the relevant Aboriginal groups is poor. This includes knowledge of both the range and quantities of foods eaten, as well as of food preparation methods (eg cooking methods).
- There is little available information on radionuclide uptake by terrestrial animals and plants. This pathway becomes particularly important once rehabilitation of minesites occurs.

This project aims to address these issues. Due to the large range of bushfoods eaten, a staged approach has been taken. Initially, work concentrated on consultation with Aboriginal people. This was followed by general observations of bushfood consumption. In addition, some studies of bioaccumulation by edible plants have been undertaken, primarily because of the importance of these in terms of rehabilitation.

Rehabilitation issues

The Commonwealth Government and the NT Government have agreed upon conceptual plans for rehabilitation of the Ranger Uranium Mine (RUM) site when mining has ceased. The major objective relevant to revegetation is:

> To revegetate the disturbed sites of the Ranger Project Area with local native plant species similar in density and abundance to that existing in adjacent areas of Kakadu National Park, in order to form an ecosystem the long-term viability of which would not require a maintenance regime significantly different from that appropriate to adjacent areas of the Park.

After the rehabilitation of the RUM site there will be radiological protection issues associated with the land use expectations by local Aboriginal people. Local people may want to use the abandoned mine areas for hunting and gathering, if there is food accessible and available. Of the nineteen species that have currently been flagged as rehabilitation plants (RUM Rehabilitation #26 2001) seventeen have direct uses for local Aboriginal people. These uses range from using the wood for digging sticks and didgeridoos to the harvesting of fruit from fruit trees. Therefore the planting of some of these species will encourage the occupation and use of the rehabilitated contaminated sites by local Aboriginal people for various periods of time.

In Momega (western Arnhem Land), Altman (1984) identified over 80 floral species used as bush foods, and in total 170 species of flora and fauna were observed being consumed. The foods that were chosen for initial study after discussion with local Aboriginal people in Kakadu and people who dealt with Aboriginal people on a daily basis were a number of common fruits and yams, for which no radiological data existed. These fruits and yams can play a significant role in the diet of an Aboriginal person at different times of the year.

The results in table 1 obtained for four species of common fruit will be discussed here; these are *Buchanania obovata*, *Persoonia falcata*, *Vitex accuminata* and *Syzygium eucalyptoides*.

Table 1 Fruits analysed in the Alligator Rivers Region

Scientific Name	Common Name	Gundjehmi name
Buchanania obovata	Green Plum	*Andudjmi*
Persoonia falcata	Geebung	*Andaak*
Vitex accuminata	Black Plum	*Anbalindja*
Syzygium eucalyptoides	White Apple	*Anbongbong*

Sample description, collection and preparation

After extensive consultation with and advice from local organisations it was arranged that an ***eriss*** researcher would accompany Aboriginal people from the Kakadu Family Resources Centre on food gathering trips. These expeditions began in 1997 and continued through to 2000. When on these trips, the researcher would act as a driver and an observer only, gleaning information from Aboriginal people about collection techniques, cataloguing food collected and observing methods of preparation. Information on the amount of bush food consumed, where it was gathered and the time of year was also recorded.

Buchanania obovata or *andudjmi* was collected from two sites on the Ranger Uranium Mine lease area. These fruit ripen during the pre-monsoon storm season (late Oct-Nov-Dec) or *Gunumeleng*. The first site was situated approximately 200 metres west of the tailings dam western wall and the second site in the former Jabiru East town area near the Gagadju workshop. A tarpaulin was put under the tree and then the tree was shaken, with fruit falling from the branches. The ripe fruit were then collected and bagged. These fruit are often eaten raw when collected or sometimes they are taken and pulped into a paste and then eaten.

Persoonia falcata or *andaak* were collected from the old Jabiru East town site from several different trees in the area. These fruits are available during *Gunumeleng* and *Gudjeuk* (monsoon season Jan–March). The sample collection method was identical to *Buchanania obovata*. The fruit are dried and eaten after soaking.

Vitex accuminata or *anbalindja* were collected from the East Alligator area, within the boundaries of the park ranger station and are also available during *Gunumeleng* and *Gudjeuk*. The same method was used for collection as for previous fruits. These small black fruit are eaten raw after collection and are sweet tasting.

Syzygium eucalyptoides or *anbongbong* were also collected at the East Alligator Ranger Station and are available during *Gunumeleng* and *Gudjeuk*. The ripe fruits were picked directly off the tree and are pink in colour; they are eaten raw.

All fruit samples were weighed and then oven dried at 60°C, then crushed with a mortar and pestle to a fine powder. The fruit samples are more commonly unwashed when consumed, so they were not washed when prepared for analysis. Artificial tracer isotopes of known concentration were added to the samples and the samples were then digested completely using concentrated HNO_3 and HCl. Analysis was performed by α-spectrometry. Sample sizes ranged from 2 to 10 grams, whilst detection limits were 0.1 mBq per sample.

Soil samples were taken from the base of the trees that the fruits were collected from. The samples were weighed and oven dried at 60°C, then ground to a fine powder using a ring mill prior to analysis. The sample was then cast into a resin disc and counted using low level, high-resolution γ-spectrometry as described by Murray et al (1987).

Results

Radionuclide analysis results are shown here as dry fruit concentration (Bq/kg) and as a concentration factor (CF).

The CF for a nuclide in an organism is defined as the activity of the nuclide per unit fresh weight (or wet weight) of the organism, divided by the activity of the same nuclide per unit weight of substrate (the physical medium from which the organism is assumed to obtain the radionuclide). In the present case the substrate is the soil the fruit tree grows in.

Concentration factors enable a prediction of the radionuclide concentrations which will be present in an edible food for a given substrate concentration. The concentration factor method is discussed in detail in ICRP (1978) and IAEA (1982).

In the case that site- or species-specific data are not available, default concentration factors for generic food types are often used for dose predictions. The International Atomic Energy Agency (IAEA) publishes such default values. Those for crops are based upon common agricultural crops such as above ground vegetables, leafy vegetables, root vegetables and grains and are intended to represent the edible parts at crop maturity. Default values are given in the following tables for comparison purposes, and are from IAEA (1982) Table XVII, page 64.

Radium

The ^{226}Ra concentrations for both soil and fruit were an order of magnitude higher for the *Buchanania obovata* than for any of the other fruits. This may be partly explained by the *Buchanania obovata* fruit tree location, which is situated on the Ranger Uranium mine lease area. The *Persoonia falcata* is also on the mine lease area but it is not near an access road and is surrounded by moderately thick bushland, thus protected from the dust thrown up by traffic on the unsealed road. The CFs in table 2 for ^{226}Ra for all four of the fruits are similar to the default values.

Table 2 Derived and default IAEA concentration factors for ^{226}Ra

Species	Dry Fruit Conc ^{226}Ra (Bq/kg)	Conc. Factor (CF)	Default IAEA CF
**Buchanania obovata*	17.1	0.022	0.04
Persoonia falcata	0.7	0.013	0.04
Vitex accuminata	2	0.037	0.04
Syzygium eucalyptoides	1.3	0.012	0.04

*Average of the tailings dam site and Gagadju workshop site

Polonium

For calculation of CFs for ^{210}Po, the soil concentrations have been assumed to be the same as those for ^{210}Pb. Derived CFs in table 3 for ^{210}Po in the *Buchanania obovata*, *Vitex accuminata* and *Syzygium eucalyptoides* are one order of magnitude higher than the default values. The derived CF for the *Persoonia falcata* is two orders of magnitude higher than the default CF.

Table 3 Derived and default IAEA concentration factors for ^{210}Po

Species	Dry Fruit Conc. ^{210}Po (Bq/kg)	Conc. Factor (CF)	Default IAEA CF
**Buchanania obovata*	3.7	0.007	0.0002
Persoonia falcata	3.7	0.046	0.0002
Vitex accuminata	0.5	0.004	0.0002
Syzygium eucalyptoides	1.6	0.007	0.0002

* Average of the tailings dam site and Gagadju workshop site

Uranium

Measured CFs in table 4 for ^{238}U in the *Buchanania obovata*, *Persoonia falcata* and *Syzygium eucalyptoides* compare favourably with the default values. The derived CF for *Vitex accuminata* was about five times higher than the default CF.

Table 4 Derived and default IAEA concentration factors for ^{238}U

Species	Dry Fruit Conc ^{238}U (Bq/kg)	Conc. Factor (CF)	Default IAEA CF
**Buchanania obovata*	0.6	0.001	0.002
Persoonia falcata	0.4	0.004	0.002
Vitex accuminata	0.4	0.011	0.002
Syzygium eucalyptoides	0.1	0.001	0.002

* Average of the tailings dam site and Gagadju workshop site

Order of importance

Table 5 shows, for each of the fruits studied, the relative order of importance of the radionuclides ^{210}Po, ^{226}Ra, ^{234}U and ^{238}U, in terms of radiological dose for a mine rehabilitation situation. This was calculated by multiplying the concentration factor by the dose conversion factor for ingestion of the relevant radionuclide, and then expressing the result as a percentage of the same calculation for ^{226}Ra. This calculation assumes secular equilibrium for the uranium series radionuclides in the soil, which is reasonable for mine waste rock covering a rehabilitated site.

Table 5 Radionuclide relative order of radiological importance

Species	^{210}Po	^{226}Ra	^{234}U	^{238}U
Buchanania obovata	143	100	0.9	0.8
Persoonia falcata	1571	100	5.3	4.8
Vitex accuminata	43	100	5.3	4.8
Syzygium eucalyptoides	257	100	1.1	1.0

The table shows that the general order of importance was: $^{210}Po \geq {}^{226}Ra \gg [{}^{234}U \approx {}^{238}U]$. Consequently, it will be more useful to target ^{210}Po and ^{226}Ra, rather than the uranium isotopes, in any future research studies and/or monitoring regimes on this topic.

Conclusions

Wild fruit and vegetables play an important part in a traditional Aboriginal diet, and radionuclide uptake by these foods will be particularly important for the post-rehabilitation situation for uranium mines in the Alligator Rivers Region. Unfortunately, in comparison with situation for aquatic flora and fauna, there is relatively little information available for radionuclide concentrations in local fruit and vegetables.

In this paper, data have been presented for several fruits. In terms of radiological dose for a mine rehabilitation situation, ^{210}Po and ^{226}Ra were found to be of greater importance than the uranium isotopes. The local values for concentration factors are sometimes up to two orders of magnitude different from the IAEA temperate environment default values. These results highlight the need to use local values wherever possible.

Other important factors that have emerged include food preparation and consumption habits of Aboriginal people, as they are quite different from European techniques and could potentially affect radionuclide intake estimates. This is difficult information to obtain because of the intrusive nature of such studies, and must be undertaken over a number of years.

At present, work on this project is concentrating on analysis of yam samples.

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ENVIRONMENTAL IMPACT OF MINING

Erosion and Hydrology

Temporal trends in erosion and hydrology for a post-mining landform at Ranger Mine

DR Moliere, KG Evans & GR Willgoose[1]

Background

An important part of rehabilitation planning for mines is the design of a stable landform for waste rock dumps or spoil piles, at the completion of mining, which minimise erosion and environmental impact offsite. To successfully incorporate landform designs in planning it is useful to predict the surface stability of the final landform using erosion and landform evolution modelling techniques (Evans et al 1998). Previous studies by Willgoose and Riley (1998) and Evans et al (1998) have used the landform evolution model, SIBERIA (Willgoose et al 1989), to simulate the erosional stability of the proposed 'above-grade' rehabilitated landform at the Energy Resources of Australia Ranger Mine (ERARM), Northern Territory, for a period of 1000 y.

Willgoose and Riley (1998) and Evans et al (1998) used input parameter values derived from data collected from areas of the waste rock dump (WRD) at Ranger and these input parameter values were assumed to remain constant throughout the period that was simulated (1000 y). In other words, changes in erosion rate resulting from processes such as soil and ecosystem development and armouring on the landform surface were not simulated.

Studies such as Jorgensen and Gardner (1987), Ritter (1990) and Loch and Orange (1997) confirm and describe the change in erosion rates on reclaimed mine sites with time. Therefore, in this study an attempt has been made to quantitatively assess how erosion and hydrologic characteristics of landforms, and hence SIBERIA parameter values, are affected by temporal change.

Study site

The Ranger Mine lease boundary is located in the wet-dry tropics of the Northern Territory, Australia, and is surrounded by the World Heritage-listed Kakadu National Park (see map on page 4). ERARM is approximately 230 km east of Darwin and 10 km east of Jabiru (see map on page 4). The average annual rainfall for Jabiru is approximately 1483 mm (Bureau of Meteorology 1999) and most of the rainfall in the region occurs during a distinct Wet season from October to April.

Data were collected from the following sites. These sites were considered to be representative of the surface hydrology and erosion characteristics that would exist on the WRD at Ranger mine at various stages after rehabilitation.

1 Associate Professor Garry Willgoose, Department of Civil, Surveying and Environmental Engineering, The University of Newcastle, University Drive, Callaghan, NSW 2308.

1 An area on the batter slope on the WRD at Ranger mine (WRD_0) (Evans et al 1998) was assumed to represent the proposed final rehabilitated condition immediately after mining is completed.

2 Two areas on the WRD at the abandoned Scinto 6 uranium mine (Moliere 2000) (see map on page 4) — one with concentrated flow conditions (WRD_{50C}) and the other with sheet flow conditions (WRD_{50S}) — were assumed to represent the surface condition at Ranger mine 50 years after rehabilitation.

3 Two natural, undisturbed channelised catchments in sparse, open woodland at Tin Camp Creek (Riley 1994) (see map on page 4), $Nat_{C(1)}$ and $Nat_{C(2)}$, were assumed to represent the Ranger mine surface condition after rehabilitation in the long term under concentrated flow conditions.

4 A natural, undisturbed area of bushland approximately 50 m south of Pit 1, Ranger mine, (Nat_S) (Bell & Willgoose 1997), was assumed to represent the surface condition at Ranger mine after rehabilitation in the long term under sheet flow conditions.

The batter slope on the WRD would initially be constructed during rehabilitation as a planar surface with sheet flow conditions. In time the batter slope could have two evolutionary paths (fig 1): (1) the surface will remain planar under sheet flow conditions, and (2) the surface will become incised under concentrated flow conditions.

If, as time passes after rehabilitation, runoff is directed away from the batter slope and the surface only receives direct rainfall, the site should retain a planar surface with sheet flow conditions. As soil and ecosystem development occurs at Ranger mine in the short term, the WRD_0 surface condition will change towards that similar to the surface condition of WRD_{50S}. In the long term, the surface condition of the landform at Ranger mine will evolve to that similar to the surface condition of Nat_S (fig 1).

However, if overland flow from the upper WRD surface breaches the bund at the top of the batter slope and flows over the surface then channelised flow will occur and a gully may develop. As soil and ecosystem development occurs at Ranger mine in the short term under concentrated flow conditions, the WRD_0 surface condition will change towards that similar to the surface condition of WRD_{50C}. In the long term the surface condition of the landform at Ranger mine will evolve to that similar to the surface condition of Nat_C (fig 1).

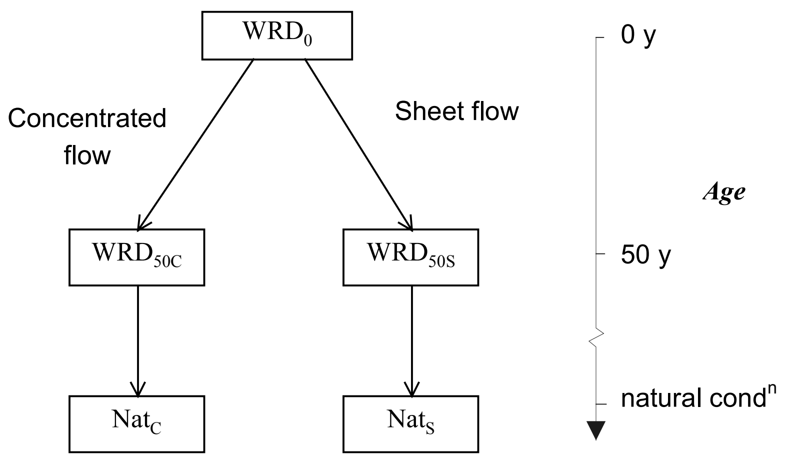

Figure 1 A schematic representation of the evolutionary paths of the batter slope at Ranger mine under concentrated flow and sheet flow conditions

SIBERIA model

The SIBERIA landform evolution model developed by Willgoose et al (1989) is a computer model designed for examining the erosional development of catchments and their channel networks.

Figure 2 shows a flow diagram of the parameter derivation process for input into the SIBERIA model. Parameter values used in SIBERIA can be classified as *primary* and *secondary*. The primary parameters in SIBERIA modelling represent the hydrologic and erosion characteristics of the site where monitoring data are collected. The secondary parameters are dependent on the primary parameter values fitted for a site and represent the long-term average SIBERIA model parameter values for the landform being modelled.

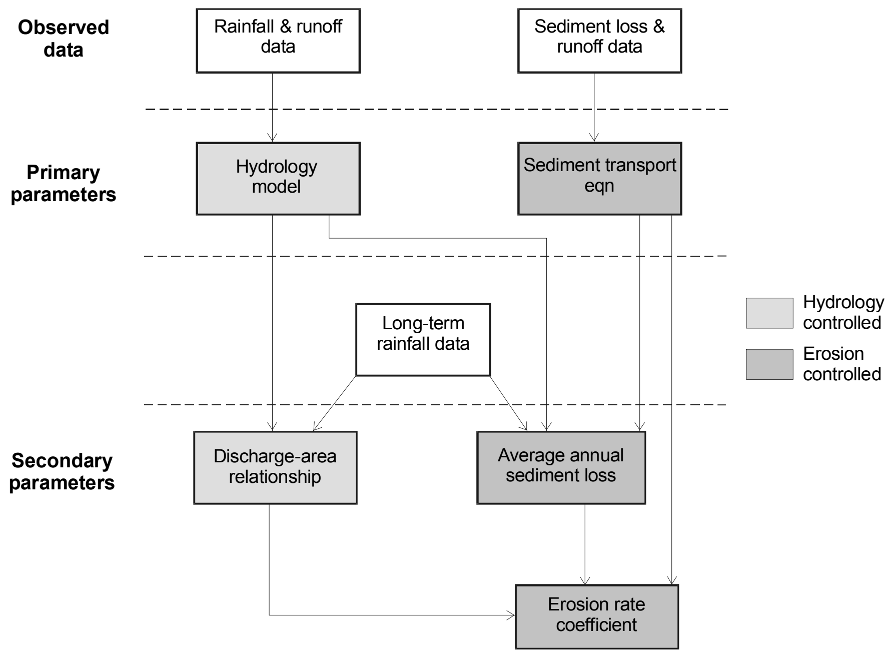

Figure 2 Flow diagram representing the SIBERIA input parameter derivation process. The shaded boxes indicate parameters that are hydrology or erosion controlled.

To obtain the primary parameter values field monitoring data are required to (1) calibrate a hydrology model using rainfall and runoff data from field sites, and (2) fit parameters to a sediment transport equation using sediment loss and runoff data from field sites, as described by Willgoose and Riley (1998), Evans et al (1998) and Moliere (2000).

Using long-term rainfall data for the region, the calibrated hydrology and erosion models for each study site (primary parameters) are used to derive long-term average SIBERIA model parameter values for the landform being modelled (secondary parameters) which, for this study, is the proposed rehabilitated landform at Ranger mine.

Temporal effect on SIBERIA model parameter values

Parameters fitted to the hydrology model and the discharge-area relationship (fig 2) represent the hydrological characteristics of the landform. These hydrology controlled parameter values fitted for each study site condition were all similar (Moliere 2000). This indicates that the model will simulate no significant change in the hydrological characteristics of the landform at Ranger mine with time.

However, it has been demonstrated that the parameter values that represent the erosion characteristics of the landform — those fitted to the sediment transport equation, the average annual sediment loss and the erosion rate coefficient (fig 2) — will change in time (fig 3). The change is rapid and occurs within the first 50 years after mining is completed, after which the parameter values return to near that of the natural landform under both concentrated flow and sheet flow conditions (fig 3). The temporal trend in these parameter values reflect the change in erosion rate with time likely to occur on the landform at Ranger mine due to factors such as compaction, surface armouring and soil and ecosystem development. It is important to incorporate the 'short-term' change in these input parameter values within landform evolution modelling to better predict the stability of the rehabilitated landform at Ranger mine for a 1000 y simulation period.

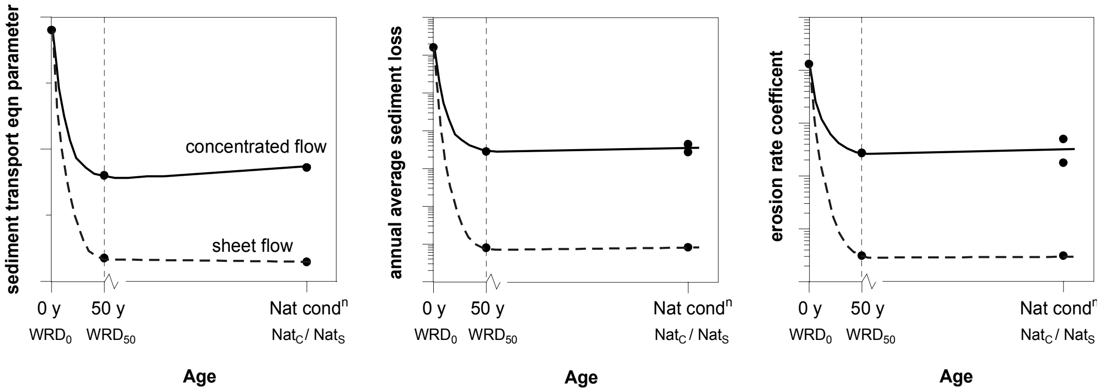

Figure 3 Erosion rate parameter values at various times after rehabilitation at Ranger mine. Approximate lines of best fit for the concentrated flow (bold) and sheet flow (dashed) conditions are also shown.

The landforms predicted at 1000 y using parameter values that change with time are shown in figure 4 and provide a current 'best estimate' of the stability of the proposed landform at Ranger mine in the long term under two different flow conditions.

SIBERIA modelling was also used to predict the valley development on the landform at 1000 y using parameter values fitted for the zero year surface condition (WRD_0). Using parameter values fitted for the WRD_0 condition it was assumed that, similar to previous studies at Ranger mine (Willgoose & Riley 1998, Evans et al 1998), there was no soil and ecosystem development on the surface of the landform and therefore the parameter values would remain constant for the simulation period (1000 y). This is a 'worst-case' scenario for the rehabilitated landform at Ranger mine (fig 4).

The difference between the overall valley development on the two best estimate landforms (fig 4) reflects the difference in erosion rate parameter values at 50 y after rehabilitation (fig 3). Using parameter values fitted for the zero year surface condition to model the rehabilitated landform, and assuming no soil and ecosystem development for the simulation period, will over-predict the long-term stability of the landform at Ranger mine, particularly under sheet flow conditions. Therefore, the worst-case scenario for the landform at 1000 y is a conservative prediction of landform stability at Ranger mine.

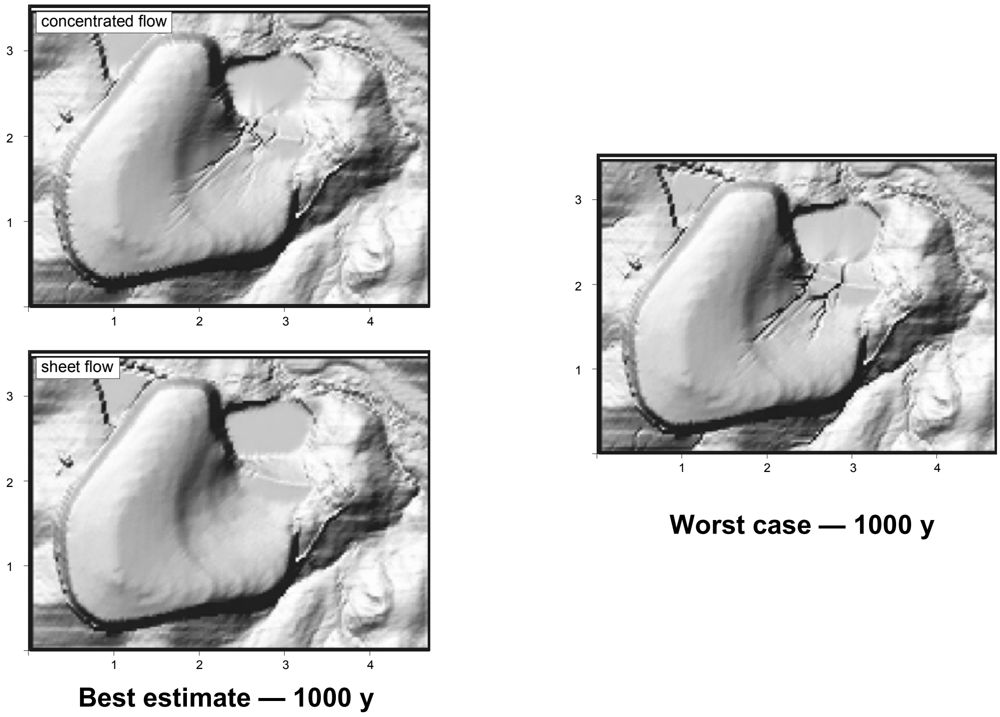

Figure 4 3-D representation of the valley development at Ranger mine at 1000 y using parameter values that (1) change with time under both concentrated and sheet flow conditions (best estimate); and (2) were fitted for the zero year surface condition (worst-case). Dimensions are in kilometres.

Conclusions

The erosion rate on the proposed landform at Ranger mine is likely to change with time over the long term and this study has been able to quantify these changes in terms of SIBERIA parameter values. The incorporation of these temporal changes in parameter values into the SIBERIA model has provided a best estimate of the stability of the landform at Ranger mine over a 1000 y simulation period. This is a significant advance in landform evolution modelling of a post-mining landform.

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Suspended sediment loads in the receiving catchment of the Jabiluka uranium mine site, Northern Territory[1]

DR Moliere, KG Evans, MJ Saynor & WD Erskine[2]

Introduction

The Jabiluka uranium mine is located in the catchment of Ngarradj[3] in the wet-dry tropics of the Northern Territory, Australia (fig 1). Ngarradj is a major downstream right-bank tributary of Magela Creek, which flows directly into the Magela Creek floodplain. The Magela Creek and floodplain are listed as Wetlands of International Importance under the Ramsar Convention and recognised under the World Heritage Convention.

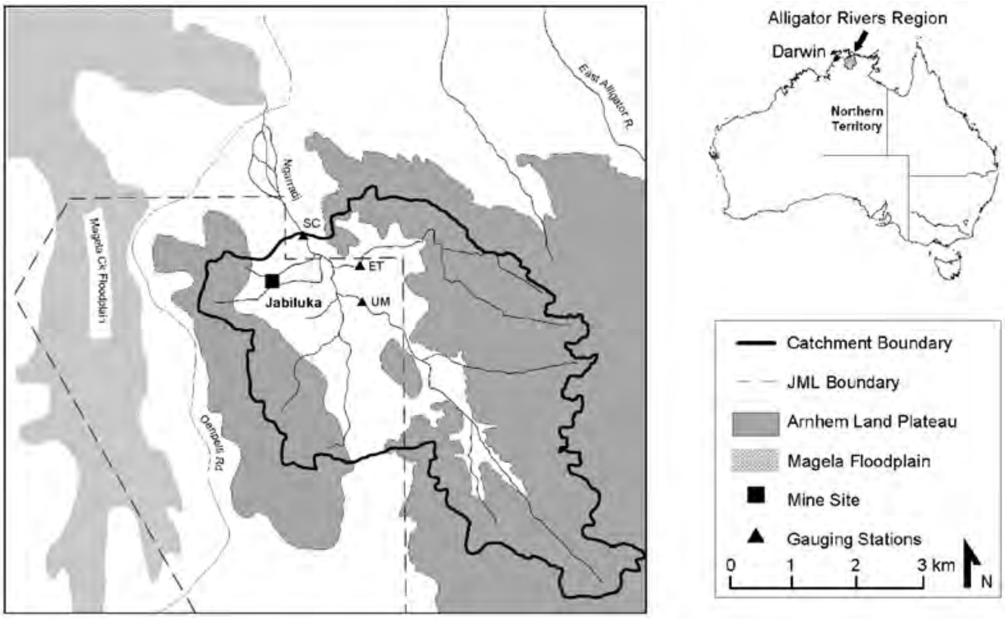

Figure 1 The Ngarradj catchment and tributaries showing the Jabiluka Mineral Lease and the gauging station sites (prepared with assistance from G Boggs, NTU)

The Ngarradj catchment will be the first to be affected should any impact occur as a result of mining operations at Jabiluka. Responsible catchment management and mining impact control requires an understanding of contemporaneous catchment baseline conditions of sediment movement and hydrology. Therefore, a stream gauging network was established in 1998 to collect data on discharge and sediment transport in the Ngarradj catchment (fig 1).

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[1] This paper appears in *Proceedings of the Hydro 2000 Conference on Interactive Hydrology*. Institution of Engineers, Perth, November 2000, 564–569.

[2] Environmental Management & Forest Practices Directorate, State Forests of NSW, Locked Bag 23, Pennant Hills, NSW 2120 (now Office of the Supervising Scientist).

[3] Ngarradj: Aboriginal name for the stream system referred to as ‘Swift Creek’ in earlier studies. Ngarradj means sulphur crested cockatoo. The full term is Ngarradj Warde Djobkeng. Ngarradj is one of several dreaming (Djang) sites on or adjacent the Jabiluka mine lease (A Ralph, Gundjehmi Aboriginal Corporation 2000).

Stream gauging stations were installed upstream (Upper Main — UM; East Tributary — ET) and downstream (Swift Creek — SC) (fig 1) of the mine in order to assess before and after impact. A specific aim of this work is to calibrate an erosion model that can be used for long-term integrated catchment management of the Jabiluka Mineral Lease with respect to suspended sediment transport.

This paper presents the calibration process of a sediment transport equation for the SC gauging site that can be used to derive a reliable and statistically significant prediction of suspended sediment loss.

Ngarradj data

Stream discharge was derived from observed stage height and velocity-area gauging data collected during the 1998/1999 and 1999/2000 Wet seasons. A stage activated pump sampler was installed at each station to obtain detailed time series variations in sediment concentrations required for accurate load determinations (Rieger & Olive 1988).

During the 1998/99 Wet season the suspended sediment data set collected at the SC gauging station was not sufficient to determine a total annual suspended sediment loss. Gaps in the suspended sediment data set were particularly evident during the periods of intense rainfall-runoff events because the capacity (ie number of bottles) of the pump sampler was exceeded (fig 2).

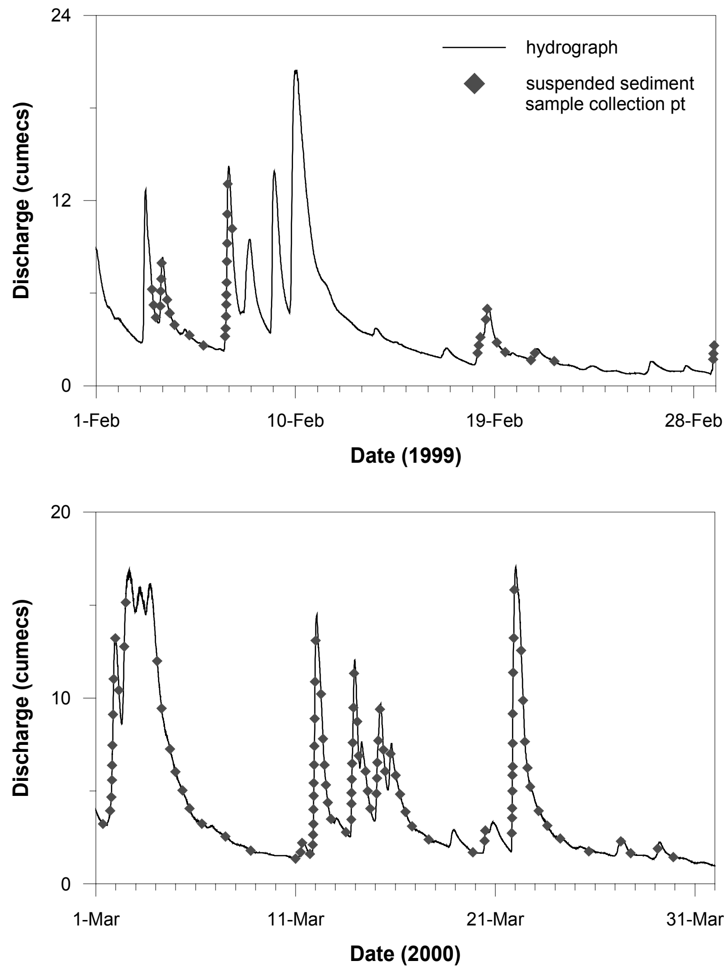

Figure 2 Hydrograph and suspended sediment sample collection points (♦) during intense rainfall-runoff periods in February 1999 (Top) and March 2000 (Bottom) at SC (Swift Creek)

In order to collect continuous suspended sediment data throughout the Wet season hydrograph at the SC gauging station, a second stage activated pump sampler was installed before the 1999/00 Wet season.

As a result, the suspended sediment data set collected during the 1999/00 Wet season at the SC gauging station was sufficient to determine a total annual suspended sediment loss. Figure 2 illustrates the hydrograph showing collection points of the sediment samples during an intense rainfall-runoff period in March 2000, indicating significant improvement in the suspended sediment data set for 1999/00 compared to that collected during 1998/99. This type of sampling distribution was achieved throughout the 1999/00 Wet season. Integration of the sedigraph gave the total observed suspended sediment loss for the 1999/00 Wet season at the SC station as 1179 t.

Sediment transport equation parameterisation

Observed sediment concentration-discharge data from SC (fig 3) were used to fit the following relationship:

$$c = K_2\ Q^{m_2} \qquad (1)$$

where c is suspended sediment concentration (g/L), Q instantaneous discharge (L/s) and m_2 and K_2 are fitted parameters.

The fitted c-Q relationship for the data collected during both 1998/99 and 1999/00 wet seasons at SC is:

$$c = 0.0056\ Q^{0.208} \qquad (r^2 = 0.03) \qquad (2)$$

which is not significant as indicated by the very low correlation coefficient (eqn 2).

Using the runoff data collected at SC during 1999/00 the fitted c-Q relationship (eqn 2) predicts a total suspended sediment loss for the Wet season at the gauging station of 1077 t, which is 91% of the observed suspended sediment load (1179 t). However, when equation 2 is corrected for statistical bias (Ferguson 1986), the predicted total suspended sediment load is 1605 t, which is not similar to the observed suspended sediment load (1179 t).

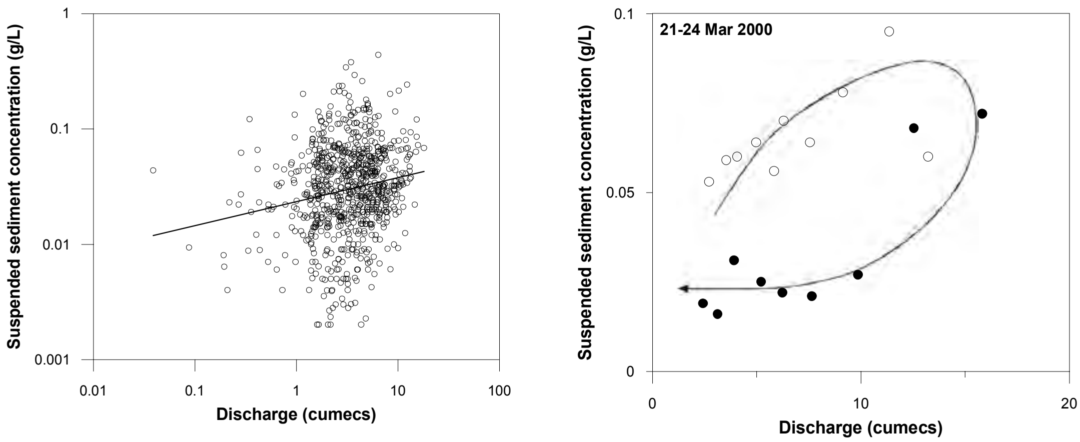

Figure 3 Suspended sediment concentration against discharge at the SC gauging station using (1) two years of monitoring data (the line of best fit (eqn 2) is also shown) (Left); and (2) monitored data from a single runoff event (21–24 March, 2000) where data were collected during the event on both the rising limb of the hydrograph (indicated by ○) and the falling limb of the hydrograph (●) (Right).

The large scatter associated with the data (fig 3) can be partly explained by the hysteresis between c and Q that occurs during individual storm events monitored on a catchment scale (Walling 1974, 1977). Hysteresis was observed in the SC data (fig 3) where the ratio c/Q at any time on the rising limb of the hydrograph is greater than that for the same discharge on the falling limb, introducing a temporal effect in the c-Q relationship (eqn 2). The 'clockwise loop' fitted for the individual storm event data (fig 3) has been attributed to sediment depletion before the runoff has peaked (Williams 1989).

Parameter values were fitted to an 'event-based' sediment transport model of the form (Evans et al 1998):

$$T = K_1 \int Q^{m_1}\, dt \qquad (3)$$

where T = total sediment loss (g), derived by integration of the sedigraph, and $\int Q^{m_1}\, dt$ = cumulative runoff over the duration of an event (Q = discharge (L/s)). Further research is required to confirm whether this removes temporal effects. Evans et al (1998) used log-log regression to fit equation 3. However, in this study a weighted regression was used where parameters K_1 and m_1 were fitted by trial and error to get a best fit slope of 1 between predicted and observed sediment loss with minimum variance (σ^2).

The monitored runoff and suspended sediment data at SC for both 1998/99 and 1999/00 Wet seasons was sub-divided into 14 discrete events. An event was considered to be a runoff period that started and ended at approximate baseflow conditions and incorporated complete rising and falling stages of the hydrograph. For example, the runoff period between 11–17 March 2000 and the runoff period between 21–24 March 2000 (fig 2) were considered to be two separate runoff events. The total runoff and total observed suspended sediment loss at the SC gauging station for each of the 14 events are given in table 1.

Table 1 Monitored event data at the SC gauging station during 1998/99 and 1999/00

Date	Total runoff (ML)	Suspended sediment loss (t)	Date	Total runoff (ML)	Suspended sediment loss (t)
9 Dec 1998	22.67	0.35	15 Jan 2000	1521.07	34.22
12 Jan 1999	273.70	8.76	23 Jan 2000	1499.18	49.51
25 Jan 1999	401.40	19.40	9 Feb 2000	5051.49	181.69
22 Mar 1999	299.75	10.76	26 Feb 2000	6079.75	278.95
20 Dec 1999	522.68	16.39	11 Mar 2000	2811.65	91.21
26 Dec 1999	2270.60	86.73	21 Mar 2000	1574.00	65.11
6 Jan 2000	1217.10	20.50	24 April 2000	768.75	17.73

An arbitrary value of m_1 was selected and used to determine a value for $\int Q^{m_1}\, dt$ for each event, which was used in equation 3 for regression analysis. The values of K_1 and m_1 were changed by trial and error until the slope of the best fit line of the linear regression between predicted suspended sediment loss (eqn 3) and observed suspended sediment loss (table 1) was equal to 1 and σ^2 was at a minimum. The values of m_1 and K_1 for this condition were selected as the fitted parameters.

This resulted in the following sediment transport equation for the SC gauging station:

$$T = 0.00136 \int Q^{1.38}\, dt \ (r^2 = 0.98;\ \text{no. of obs} = 14;\ p < 0.001) \qquad (4)$$

Figure 4 shows that the predicted suspended sediment losses (eqn 4) are similar to the observed sediment losses for the 14 runoff events at the SC gauging station. Equation 4 is a statistically significant and reliable sediment loss-discharge relationship for this site. The total predicted suspended sediment loss at the SC gauging station using the observed hydrograph for the 1999/00 wet season in equation 4 is 1133 t, which is 96% of the observed suspended sediment loss of 1179 t.

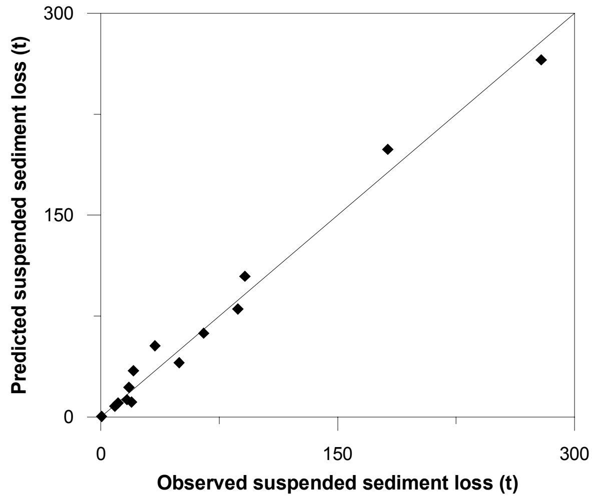

Figure 4 Relationship between observed and predicted suspended sediment loss (eqn 4) at the SC gauging station

Conclusion

In this study, a more reliable and statistically significant prediction of suspended sediment loss at the SC station is obtained using an event based total sediment transport equation instead of a *c-Q* relationship. The total sediment transport equation predicts waves of sediment which should remove the temporal hysteretic effect within individual runoff events.

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Integration of GIS and modelling techniques for impact assessment at Jabiluka Mine, Northern Territory[1]

GS Boggs[2,3], KG Evans, CC Devonport[2] & MJ Saynor

Introduction

This section describes the development and application of a geographic information system (GIS) centred approach to risk assessment that is being used to measure the long-term geomorphological impacts of the Environmental Resources of Australia (ERA) Jabiluka Mine. GIS provides a means by which the data collected during the assessment of possible mining impacts can be stored and manipulated. geographic information systems have been linked with a large number of erosion and hydrology models (de Roo 1998) and used in a limited number of geomorphological impact assessment studies (eg Patrono et al 1995). However, this study differs from many previous studies by adopting a GIS centred approach to the management and manipulation of data generated by a geomorphological impact assessment. The design of the GIS is based on three major functions; (1) data storage, management and retrieval, (2) interaction with environmental modelling techniques, and (3) basin analysis including the geomorphometric analysis of landform evolution (fig 1). Benefits of this approach include the simplification of data maintenance, revision, and update, as well as increasing the accessibility and useability of the data.

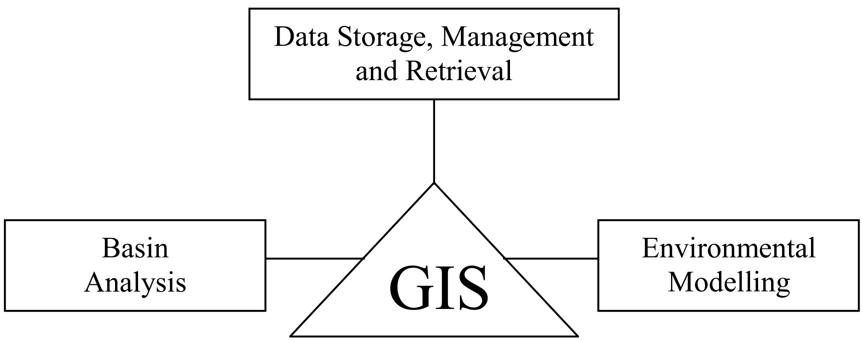

Figure 1 The design of a GIS as a central point for risk assessment

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1 More detailed discussion of this research is provided in Boggs GS, Devonport CC, Evans KG, Saynor MJ & Moliere DR 2001. *Development of a GIS based approach to mining risk assessment*. Supervising Scientist Report 159, Supervising Scientist, Darwin.

2 GIS, Faculty of Science & Education, Northern Territory University, Darwin NT 0909.

3 Present address: Department of Geography, The University of Western Australia, Crawley WA 6009.

Data storage, management and retrieval

Gauging station data

In November 1998 three stream gauging stations were established within the Ngarradj catchment (fig 2). Data are collected from these stations at frequent intervals, providing a high temporal resolution data source. These data are being used to: (i) monitor for potential impacts of the ERA Jabiluka Mine (Erskine et al 2001) and (ii) to calibrate the SIBERIA landform evolution model (Boggs et al 2000). The approach adopted to store, manage and retrieve these data involved the customisation of the ArcView GIS package to connect the GIS with the Microsoft Access relational database and Microsoft Excel spreadsheet software packages (Boggs et al 1999). The system has been designed to enable data collected during field visits or through laboratory analyses to be entered into an access database linked to the GIS. This allows data to be retrieved through customised dialog boxes embedded within the GIS interface.

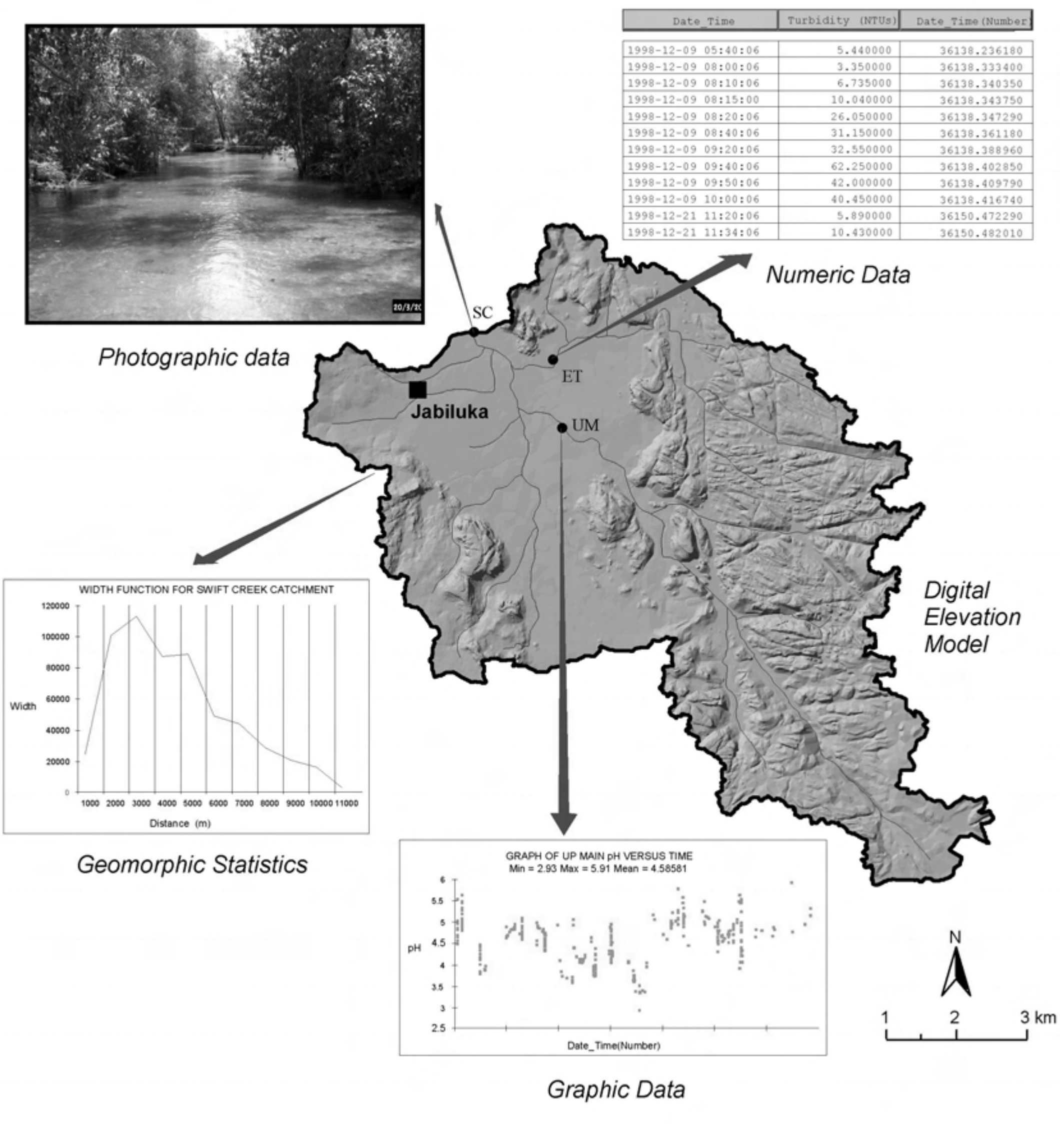

Figure 2 A hillshaded version of the Ngarradj digital elevation model, showing types of data that can be rapidly accessed through the customised GIS interface (SC, ET & UM are stream gauging stations)

The link between the GIS and database allows the user to select a site on the computer screen and interact with the associated databases, importing and graphing data ‘on the fly’ for the selected site (fig 2). This allows rapid assessment of temporal and spatial trends in the data. Much of the analysis of these data, however, involves complex statistical operations not offered within the standard GIS. An option within the data retrieval dialog box exports data directly from the database to a spreadsheet package, allowing further statistical analysis and more sophisticated graphing operations.

Raster data

Raster data obtained for the Ngarradj project comprise a Digital Elevation Model (DEM), constructed from 1:30 000 pre-mining (1991) aerial photography (fig 2) and remotely sensed imagery (including aerial photography, Landsat TM and MSS imagery). Raster data sets generally utilise large amounts of storage space and thus require large and well-organised data bases. A GIS offers a highly suitable approach for efficient storage, retrieval and analysis of large raster data sets (Schultz 1993). DEMs are currently used in many geomorphologic studies as they allow the extraction of terrain and drainage features to be fully automated (Moore et al 1991). Remotely sensed imagery is considered to be a rapid and flexible method for obtaining updated data, particularly as images are easily stored and interpreted in a GIS. Raster data, therefore, are useful in the examination and explanation of the gauging station data and provide direct inputs into the hydrology and landform evolution models.

Vector data

The vector database established consists primarily of the Topo-250 k digital data product produced by the Australian Surveying and Land Information Group (AUSLIG), with some of the data available at 100 k scale. Additional data layers are related to individual projects and have been obtained in the field or from aerial photography and other imagery. As much of the base vector data are too coarse for investigations at the Jabiluka project scale, the primary vector data source is derived from DEM/remote sensing products and data collected with the use of a differential GPS (Global Positioning System). Differential GPS provides a cost effective, accurate source of raw geographical information valuable in the mapping, field data collection and GIS database construction phases of the geomorphological impact assessment process (Cornelius et al 1994). Differential GPS, along with aerial photography interpretation, has been successfully used in initial channel reach characterisation and geomorphic mapping of the Ngarradj (Swift Creek) catchment. In a project of this scale, the use of a differential GPS to geo-referenced field sites is considered crucial to the GIS-centred data management approach as all data are linked to specific spatial locations, facilitating the incorporation of all field project data into the information management system.

Geomorphological modelling with GIS

Currently three environmental models are employed in the assessment of mine site landform stability and off-site geomorphological and environmental impacts: 1) a basic sediment transport model, 2) the Distributed Field Williams (DISTFW) hydrology model (Field & Williams 1987), and 3) the SIBERIA landform evolution model (Willgoose et al 1991).

The sediment transport model is an equation that does not have a spatial component and is therefore not appropriate for implementation within a GIS. However, the DISTFW hydrology model is a distributed model that operates on a sub-catchment basis, whilst the SIBERIA landform evolution model is based on a DEM. The integration of these two models with the GIS has used the loose coupling and tight coupling methods respectively. The ArcView GIS package has been customised to facilitate these levels of integration between the models and the GIS.

DistFW hydrology model

Hydrologic analysis has been integrated with computers to such an extent that computers often provide the primary source of information for decision-making by many hydrologic engineers (deVantier & Feldman 1993). Linking the DISTFW hydrology model with a GIS using a loose coupling approach primarily involves the development of a GIS toolbox that will allow the automatic generation of DISTFW input requirements.

The DISTFW hydrology model requires the input of a significant amount of topographic information. Catchments are represented within the model as a number of sub-catchments for which information must be derived describing their horizontal shape, vertical relief, conveyance and the flow relationships between individual sub-catchments. A significant challenge in this research project has been to develop a set of customised tools that automatically generates this information primarily from a DEM. Six software tools have now been developed that extend the functionality of the ArcView GIS to satisfy the topographic input requirements of the DISTFW hydrology model (Boggs et al 2000).

SIBERIA landform evolution model

SIBERIA models the evolution of a catchment through operations on a DEM for the determination of drainage areas and geomorphology. GIS offer a wide range of raster data processing capabilities and a clear means for organising and visualising data from a number of different formats (Rieger 1998). Linking the SIBERIA landform evolution model with GIS therefore provides benefits not available in one or other of these environments. In particular, integrating the SIBERIA model with a GIS enhances the 'user-friendliness' and functionality of the model, which would not otherwise lend itself to interactive use. The approach adopted to link SIBERIA with a GIS maintains the two technologies as separate entities that share a user friendly front end and database.

The suite of tools developed to link SIBERIA with the ArcView GIS package have been assembled into an ArcView extension named 'ArcEvolve'. Extensions are add-on programs that provide additional functionality to ArcView through the addition of menu items, buttons and/or tools. The functionality associated with the added menus/button/tools is derived from scripts written in the ArcView object-oriented programming language 'Avenue'.

ArcEvolve allows the user to interact with SIBERIA through the addition of a menu ('SIBERIA') to the ArcView 'View' document graphical user interface (GUI). The menu contains a number of items that:

- allow SIBERIA native format files to be imported and exported. The digital elevation model data and parameter values contained in a SIBERIA restart file (rst2) can be imported into an ArcView grid and associated database respectively. SIBERIA boundary files, which contain information for an irregularly shaped region can be imported or created from an ArcView grid.
- provide access to dialog boxes for the creation and management of a SIBERIA parameter database. A new database can be created and parameters imported, with each record being linked to a grid. A series of nine dialog boxes, which can be accessed from the View GUI, provide a user-friendly frontend for updating the parameter database for a selected grid.

- Run the SIBERIA model, with output imported into ArcView following the completion of the model run

Basin analysis

Geomorphometry, defined as the 'quantitative treatment of the morphology of landforms', (Morisawa 1988) has expanded significantly since its inception over 50 years ago (Morisawa 1988). The advent of the DEM has allowed geomorphometry to not be limited to the time-consuming measurement of landform properties from contour lines on topographic maps. The DEM has allowed the development of algorithm's that rapidly derive such measures and has also allowed the definition of a number of new morphometric measures (Nogami 1995). The width function, hypsometric curve, cumulative area distribution and area-slope relationship are four geomorphometric measures that can be rapidly derived from a DEM and have been shown to be important measures of catchment geomorphology and hydrology. These descriptors have also been successfully used to quantify and compare SIBERIA derived landscapes with natural landscapes (Hancock et al 2002). This study represents the first attempt to apply these measures to assessing the impact of mining on catchment evolution.

A number of tools for the geomorphometric analysis of digital elevation data (the primary input and output of SIBERIA) have been developed or included as part of the ArcView extension ArcEvolve. The tools, contained in a second menu, 'Geomorph', allow geomorphic descriptors including the width function, hypsometric curve, cumulative area diagram and area-slope relationship to be calculated. The tools also allow the direct comparison of two SIBERIA output DEMs by the calculation of the denudation rate and volumetric difference between two surfaces.

Conclusions

A GIS centred approach to risk assessment has been developed that will be used for a geomorphological investigation into the impact of the ERA Jabiluka Mine on the Ngarradj catchment. A GIS based approach to data management simplifies data maintenance, revision, and update, as well as facilitating data availability and access for users. Furthermore, linking the GIS to the DIST-FW hydrology model and SIBERIA landform evolution model provides a more user-friendly approach to landform evolution modelling whilst contributing significant pre-processing and analysis capabilities to the modelling process.

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An erosion assessment of the former Nabarlek uranium mine, Northern Territory

MK Grabham[1,2], GR Hancock[1] & KG Evans

An appraisal of the rehabilitated landsurface of the former Nabarlek uranium mine was conducted to assess the site from a soil erosion perspective as part of an independent assessment commissioned to evaluate the overall success of rehabilitation of the site. Determination of the gross erosion occurring on the site, sediment discharge to Cooper Creek and the resultant concentration of sediment in Cooper Creek were the primary objectives of the study. These objectives were achieved through the application of several models which used parameter values collected from the Nabarlek site during the Dry season of 2000.

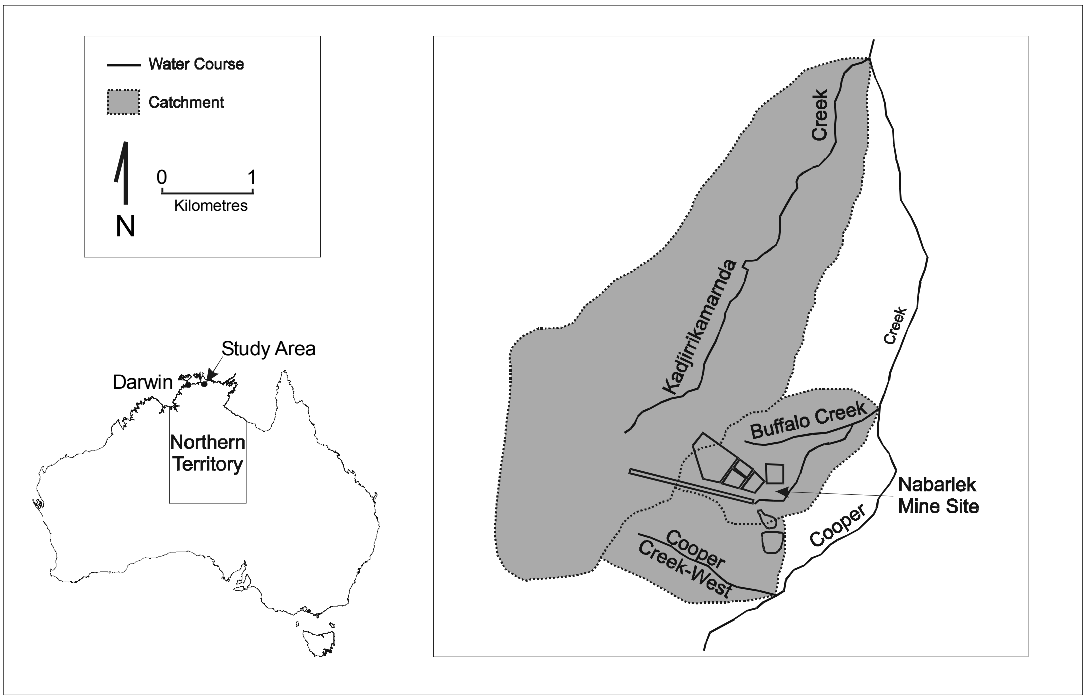

Figure 1 Location and catchments of the former Nabarlek uranium mine

The Nabarlek mine site is located 270 km east of Darwin near the western edge of Arnhem Land. History of the site has been described by Waggitt (2000). The 173 hectare site lies within the catchments of the Cooper (west), Buffalo and Kadjirrikamarnda Creeks (fig 1). These three catchments drain into Cooper Creek which in turn discharges into the mouth of the East Alligator River. Cooper Creek drains 25 240 hectares above its confluence with Kadjirrikamarnda Creek. The catchment is composed of undulating plains of red and yellow soils and siliceous sands with the upper sections of the catchment draining the escarpment areas of the Arnhem Land plateau (Galloway 1976, Needham 1982, Riley 1995). The vegetation is described as open dry-sclerophyll forests (Story et al 1976).

[1] School of Geosciences, The University of Newcastle, Callaghan, NSW 2308.
[2] Present address: NSW Agriculture, PO Box 1087, Griffith, NSW 2680.

Methods

The Revised Universal Soil Loss Equation (RUSLE) (Renard et al 1994) was used to determine the gross erosion occurring on the Nabarlek site. This model has been applied to mining applications in the Northern Territory (Riley 1994, Evans & Riley 1994, Riley 1995, Evans & Loch 1996, Evans 1997, Evans 2000) and its predecessor — the Universal Soil Loss Equation (USLE) — was used to design the final landform at Nabarlek (Riley 1995). This not only provides opportunity for monitoring the predictive success of the RUSLE, but also indicates the applicability of the model to the tropical environment of northern Australia (Riley 1995).

The RUSLE is given as:

$$A = RKLSCP \tag{1}$$

where A is the annual average soil loss (Mg ha^{-1} year^{-1}), R is the rainfall factor expressed as an erosion index (EI) (MJ mm (ha hour year)$^{-1}$), K is the soil erodibility factor (Mg ha hour (ha MJ mm)$^{-1}$), LS is the slope-length and slope-steepness factor (dimensionless), C is the cover and management factor (dimensionless), and P is the conservation practice factor (dimensionless) (Renard et al 1994, Rosewell & Loch 1996).

To successfully apply the RUSLE to a variable landscape, such as the Nabarlek site, it is necessary to divide the area up into homogeneous sub-areas (Wischmeier 1977). Twelve sub-areas or 'erosion units' were determined using the primary components of the RUSLE as the discriminating factors. The gross erosion occurring on each of these twelve units was then calculated and the results added to provide a gross sediment loss estimation for the site.

A percentage of the annual gross erosion from within a catchment arrives at the catchment outlet each year (Robinson 1977, Walling 1983). The remaining sediment is held in depositional areas of a catchment as part of the progressive cycle of detachment, deposition and re-entrainment of the eroded material. To quantify the proportion of the gross erosion released from a catchment requires the application of a Sediment Delivery Ratio (SDR).

The derivation of effective SDRs has proved illusive (Walling 1983, Naden & Cooper 1999) resulting in a lack of models with universal application (Walling 1983). Consequently an SDR developed in the Alligator Rivers Region (ARR) was applied. Evans (2000) derived the SDR for Gulungul Creek in the ARR using the catchment area data of Robinson (1977). The proximity of this location to the Nabarlek site and its ease of application makes this SDR ideal. The ratio is given as:

$$SDR = (8.33 - 0.51 \ln A)^2 \tag{2}$$

where A is the catchment area (ha).

To determine sediment concentration it is necessary to calculate discharge. Discharge can be determined using a hydrological model, of which there is a vast array (Ward & Robinson 1990). A simple model found to provide reliable mean annual discharge results for Gulungul Creek (Evans 2000) and other locations in the ARR (Duggan 1994) was implemented at the Nabarlek site. This model is expressed as:

$$Q = C_r RA \tag{3}$$

where Q is the mean annual discharge (m^3 year^{-1}), C_r is a runoff coefficient, R is the average annual rainfall (m year^{-1}) and A is the catchment area (m^2). The C_r factor is dependent on the topography, geology and size of a catchment (Ward & Robinson 1990) and was determined to be 0.43 in undisturbed areas of the catchment and 0.1 for the disturbed areas of the Nabarlek catchment.

Natural sediment loss can be calculated by using a denudation rate. As denudation rates are location specific it is necessary to use a value which has been derived from local topography. There are several values which have been determined for the ARR. These values range from 0.01 mm yr^{-1} (Airey 1983) to 0.04 mm yr^{-1} (Cull et al 1992). Erskine and Saynor (2000) determined the denudation rate for Swift Creek (Ngarradj), an area of similar topography to Cooper Creek, to be 0.016 mm yr^{-1}. This value has been used to determine the sediment yield from natural areas of the Cooper Creek catchment.

Sediment loss from the Nabarlek site was assessed under two scenarios. The first scenario assessed the site under the conditions that were present at the time of sampling. This scenario is referred to as the vegetated scenario. The second scenario assumes no vegetation cover is present thereby simulating the site under burnt conditions. This was carried out as the Nabarlek area has a fire recurrence interval of between one in every two years and one in every three years. As total removal of all vegetation cover is unlikely, this scenario provides a 'worst case' situation.

Results

RUSLE results predict that, under vegetated conditions, there is an average of 31, 335 and 133 tonnes per year of sediment removed from the Nabarlek site into the Cooper-west, Buffalo and Kadjirrikamarnda Creek catchments respectively. Under non-vegetated conditions there is an average of 227, 1238 and 190 tonnes per year of sediment removed respectively. Using a natural denudation rate of 0.016 mm yr^{-1} and catchment discharge values, the background stream sediment concentration for the streams draining Nabarlek is 33 mg L^{-1}. Background, vegetated and non-vegetated sediment concentration for the respective catchments are shown in figure 2.

The background stream sediment concentration in Cooper Creek at the mouth of the three catchments draining Nabarlek is 33 mg L^{-1} given a denudation rate of 0.016 mm yr^{-1} and the annual discharge. The change in sediment concentration as a result of sediment influx from the rehabilitated site is simply a dilution of the above mentioned catchment sediment concentrations. The estimated stream sediment concentration at each of these confluences is illustrated in figure 2.

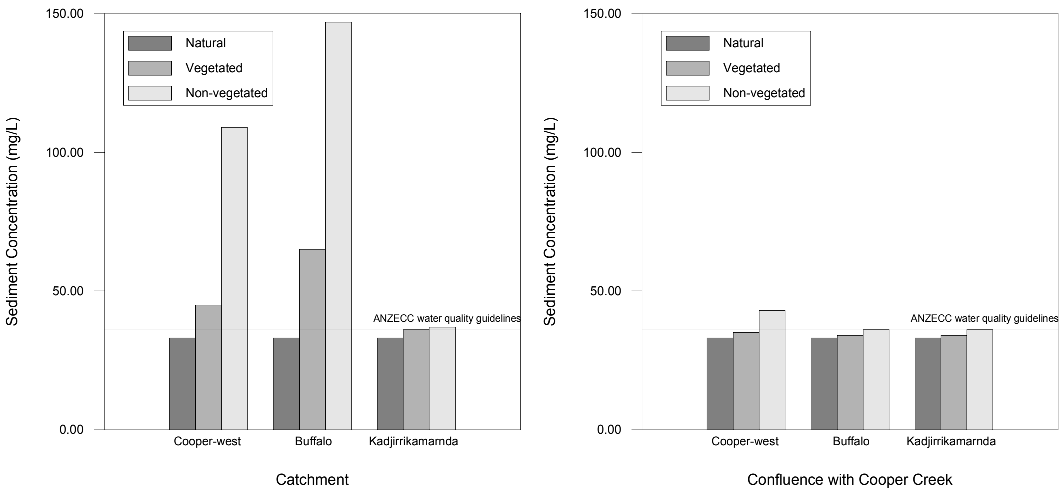

Figure 2 Sediment concentration in catchments draining Nabarlek mine site (left), stream sediment concentration in Cooper Creek at catchment confluences (right).

Discussion

Gross erosion on the Nabarlek site is higher than that occurring on natural areas of the catchment. However, the results show the site to be eroding at a rate close to that predicted by Riley (1995). This suggests that the rehabilitated landscape will maintain its integrity for the recommended design life of 1000 years (DASETT 1987). Furthermore, the average sediment concentration in Cooper Creek is generally within 10% of natural sediment concentration as per the ANZECC water quality guidelines. However, stream sediment concentrations in the catchments draining the Nabarlek site — Cooper-west, Buffalo and Kadjirrikamarnda Creeks — may exceed water quality guidelines. The high sediment concentrations predicted for these catchments were based on non-vegetated conditions and therefore represent a worst case scenario.

The sediment concentration predictions in this study failed to take into consideration the presence of sediment containment ponds which are present on some of the streams draining Nabarlek. While the initial sediment concentration will be lower as a result of sediment containment, the ponds may eventually be breached leading to the subsequent discharge of the sediment to Cooper Creek. In addition to these structures, the impact of feral animals on the site has not been taken into consideration. There is extensive evidence suggesting the presence of both feral pigs and horses on the Nabarlek site. The disturbance of the soil surface by these animals may contribute to the sediment yield from the site.

To evaluate the accuracy of the results in this report, monitoring of the site is necessary. Plot studies and the installation of devices such as erosion pins would provide adequate results to compare with the theoretical predictions in this report. The monitoring of gully development using surveying techniques would provide information on development rates and provide an effective monitoring program for the Nabarlek site.

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Sediment loss from a waste rock dump, Ranger mine, northern Australia[1]

MJ Saynor & KG Evans

Introduction

At the conclusion of mining at the Ranger Mine, it is planned to isolate tailings from the environment for at least 1000 years (East et al 1994). This will entail the construction of a suitable containment structure.

One of the major pressures on the integrity of a containment structure is erosion, causing landform instability, resulting in exposure of encapsulated contaminants, elevated sediment delivery at catchment outlets, and subsequent degradation of downstream water quality (Evans 2000). Erosion rates on containment structures can be quantified using 3-dimensional landform evolution simulation techniques (SIBERIA) (Willgoose et al 1991).

Vegetation generally reduces erosion and the effects should be quantified to enhance rehabilitation design at the Ranger mine. This has been done using large scale plot erosion (600 m^2) data from the waste rock dump to: (1) derive site-specific linear relationships between bedload and total sediment load; and (2) provide an understanding of the variation of sediment yield during the Wet season as site conditions change.

Methods

During the 1994/95 Wet season data (table 1) were obtained under natural rainfall events for two sites (soil and fire sites) on the waste rock dump at the Ranger Mine (fig 1). The sites were established in November 1994. The surface (average slope of 1.2%) of the soil site had been ripped and topsoil added approximately 8 years prior to this study and now had a vegetation cover of low shrubs and grasses (*Acacia* and *Sorghum* species). The surface of the fire site (average slope of 2.3%) was initially ripped and topsoil added, and was now vegetated with low shrubs, grasses and well established trees (*Eucalyptus*, *Acacia*, *Grevillea* species) approximately 10 years old. Both the soil and fire site had high levels of surface roughness due to the ripping and the presence of large, competent, rock fragments. Formation of debris dams also increased the surface roughness of both plots. A third site, the cap site, established in 1993, had an average slope of 2.8%, was not surface ripped, had negligible vegetation and a cover of fine surface material over a pan. This site had low surface roughness. Only rainfall, runoff and bedload data were collected on the cap site during the 1994/95 Wet season.

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[1] A more detailed discussion of this research is provided in Saynor MJ & Evans KG 2001. Sediment loss from a mine waste rock dump, Northern Australia. *Australian Geographical Studies* 39(1), 34–51.

Table 1 Fire and soil site data, for monitored events during the 1994/95 Wet season. Cap site data from Evans (1997) are also shown.

Site	Date	Total rainfall (mm)	Maximum 10 minute rainfall intensity (mm h^{-1})	Total discharge (l)	Runoff coefficient	Suspended load (g) [Concentration (g L^{-1})]	Bedload (g) [Concentration (g L^{-1})]	Total load (g) [Concentration (g L^{-1})]
Fire	17/1/95	6.2	31	121.9	0.03	9.7 [0.08]	92.4 [0.76]	102.1 [0.84]
	19/1/95	12.8	50	356.1	0.05	40.3 [0.11]	98.6 [0.28]	138.9 [0.39]
	20/1/95	7.2	41	65.9	0.02	8.6 [0.13]	43.0 [0.65]	51.6 [0.78]
	25/1/95	44.4	128	1254.6	0.05	12.0 [0.01]	248.4 [0.20]	368.4 [0.29]
	10/2/95	20.4	59	440.0	0.04	35.3 [0.08]	38.4 [0.09]	73.7 [0.17]
	18/2/95	38.2	90	1383.9	0.06	69.6 [0.05]	438.4 [0.32]	508.0 [0.37]
	28/2/95	33.4	86	853.1	0.04	65.3 [0.08]	268.2 [0.31]	333.5 [0.39]
	1/3/95	14	44	128.4	0.02	16.2 [0.13]	107.2 [0.84]	123.4 [0.96]
	27/3/95	8	37	62.8	0.01	12.1 [0.19]	13.6 [0.22]	25.7 [0.41]
Soil	17/1/95	6.6	34	288	0.07	29.8 [0.10]	118.2 [0.41]	148.0 [0.51]
	19/1/95	18.4	68	1397	0.13	111.5 [0.08]	325.0 [0.23]	436.5 [0.31]
	20/1/95	9.8	56	596	0.10	58.5 [0.10]	556.1 [0.93]	614.6 [1.03]
	25/1/95	39.4	118	2303	0.10	251.3 [0.11]	889.7 [0.39]	1141 .0[50]
	27/1/95	7.4	37	269.1	0.06	19.0 [0.07]	85.8 [0.32]	104.8 [0.39]
	10/2/95	20.4	55	1142	0.09	89.4 [0.08]	277.8 [0.24]	366.4 [0.32]
	18/2/95	48.4	121	4020	0.14	676.2 [0.17]	1357.6 [0.34]	2033.8 [0.51]
	28/2/95	28	73	1805	0.11	298.5 [0.17]	327.0 [0.18]	625.5 [0.35]
	8/3/95	12.8	74	956.1	0.12	108.1 [0.11]	234.5 [0.25]	342.6 [0.36]
	27/3/95	8	42	241.8	0.05	27.1 [0.11]	58.1 [0.24]	85.2 [0.35]
Cap	16/11/93	18	54	6509	0.61	4440 [0.68]	6074 [0.93]	10514 [1.62]
	09/12/93	49	132	21638	0.75	18930 [0.88]	25007 [1.16]	43937 [2.03]
	10/12/93	11	30	4450	0.68	721 [0.16]	814 [0.18]	1535 [0.35]
	20/12/93	9	48	3013	0.57	611 [0.20]	2767 [0.92]	3377 [1.12]
	21/02/94	16	54	9988	1.06	1683 [0.17]	2914 [0.29]	4597 [0.46]

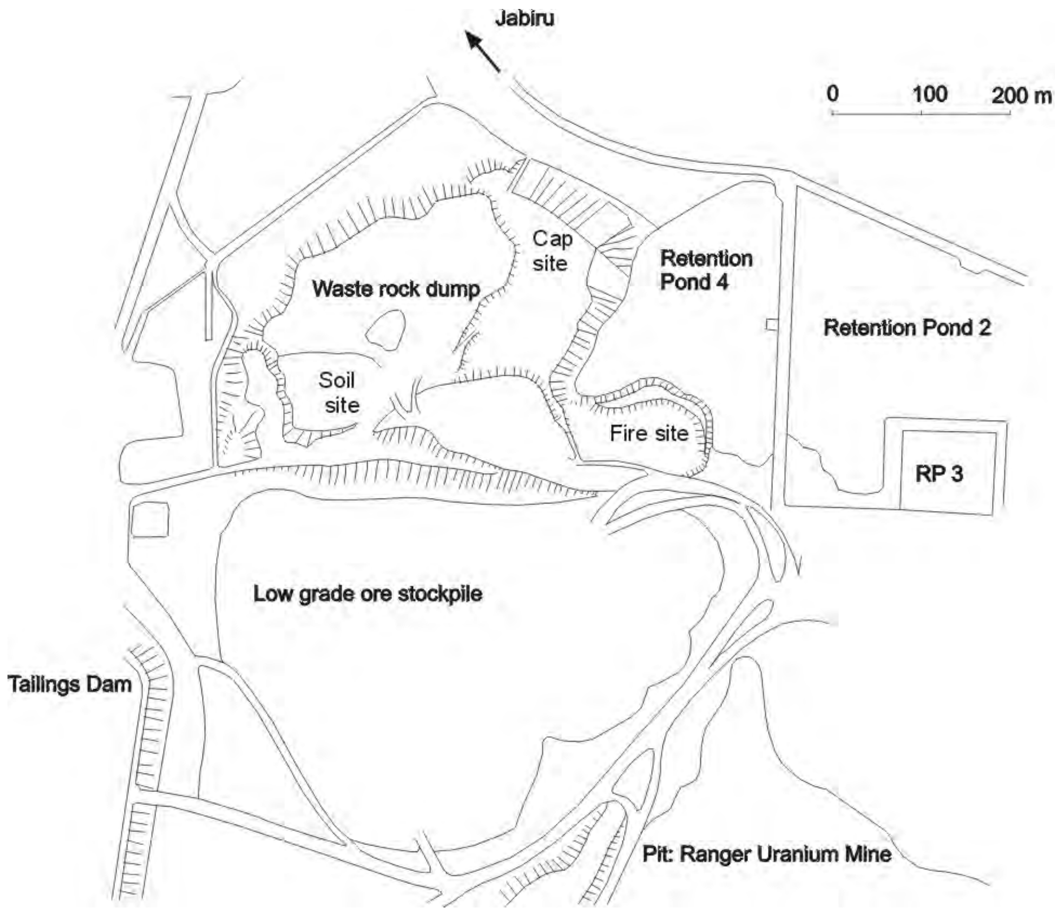

Figure 1 Location of the study sites at Ranger mine

A vegetation survey on the soil and fire sites commenced on 30/1/95 (R Hall pers comm, 1996). Representative 1 m^2 quadrants (randomly chosen) were pegged at four locations at each plot and fortnightly measurements of living ground cover (percentage and approximate mean grass heights) were made.

Bedload and total load relationships

There was a significant linear relationship between bedload and total sediment loss for both the soil and fire sites (fig 2). 1993/94 Wet season data from events observed on the cap site (Evans 1997) (table 1) also indicate a significant relationship between bedload and total sediment loss (fig 2). These relationships (fig 2) make it possible to predict the total sediment loss for a storm using collected measured bedload from the studied sites.

Bedload measurements were greatest from the unvegetated cap site and lowest from the densely vegetated fire site. The site-specific relationships between total sediment load and bedload indicate that the percentage of suspended sediment discharge from the plot decreases as the percentage plant cover increases. This may be due to a filtering effect by the vegetation. The power function

$$\text{total sediment loss} = 1.28 \times \text{bedload}^{1.02} \qquad (r^2 = 0.99,\ p<0.001) \qquad (1)$$

is applicable to all three sites. However, it under-predicted total sediment load during the two largest storms on the cap site (table 1) by approximately 14%. The cap site linear function (fig 2) over-predicted the event on 16/11/93 by 0.95% and under-predicted the event on 9/12/93 by 0.55%.

Generally, <20% of runoff events are responsible for >65% of total erosion losses (Edwards 1987, Wockner & Freebairn 1991, Erskine & Saynor 1996) making it important that high sediment loss events are well predicted. The site-specific linear equations (fig 2) are more appropriate than the power function for predicting total sediment load from bedload, particularly with respect to the larger, more damaging, events. Both the power function (eqn 1) and the fire site linear function (fig 2) under-predict sediment loss for small events. The magnitude of the sediment loss at the lower end compared with losses at the higher end, make under-prediction at the lower end much less significant than at the higher end for landform rehabilitation design.

Similar relationships have previously been derived for a site in the USA (Olyphant et al 1991). The relationships derived in this study, specific to a Northern Australian mine site, have important application in erosion modelling and monitoring, as outlined below.

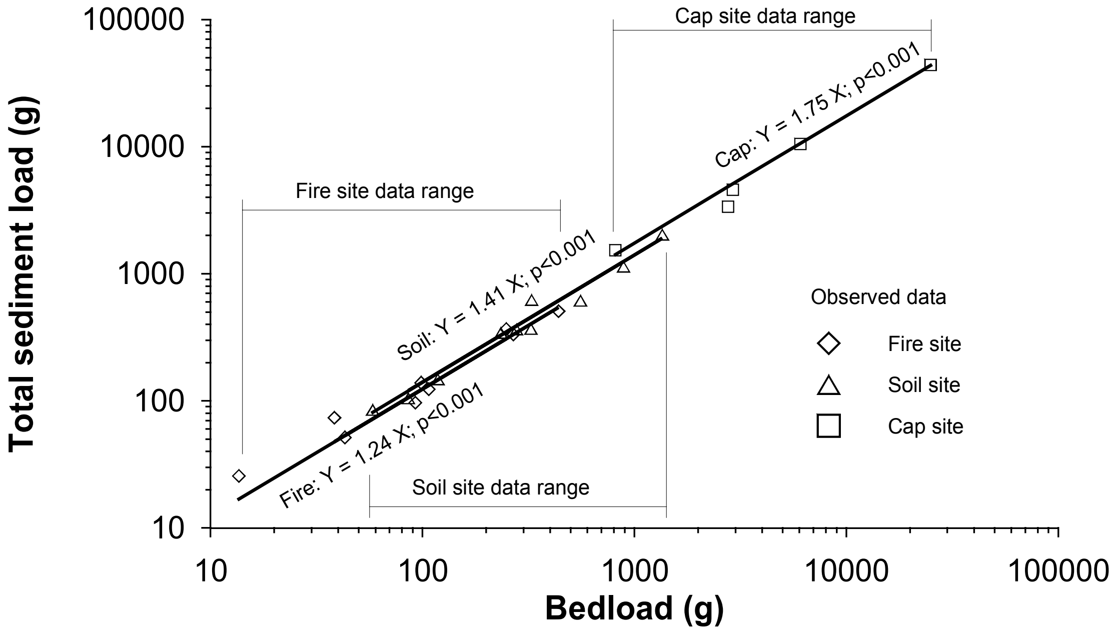

Figure 2 Fitted bedload — total sediment load relationship for the study sites and data ranges for the sites. Cap site data are from the 1993/94 Wet season (Source: Evans 1997).

Application to erosion modelling

The sediment discharge equation in the SIBERIA landform evolution model has been calibrated using a total sediment–total discharge relationship of the form:

$$T = \beta' \int Q^{m+1} dt \qquad (2)$$

where T = total sediment load (g), $\int Q^{m+1} dt$ = total discharge (L) and β' is a fitted parameter (Evans et al 1998). For this equation the total load, comprising the bedload collected at the end of an event and the total suspended load, derived through integration of sediment discharge (Q_s) (g s^{-1}), is fitted against $\int Q^{m+1} dt$. The exponent on Q, $(m+1)$ is fitted to each instantaneous discharge within the integral (Evans et al 1998). This may reduce the effect of the hysteresis identified in sediment discharge events as using total event data removes the temporal component within an event (Moliere et al 2002 — this volume).

Bedload is more easily collected than the suspended load. Therefore bedload - total sediment load relationships based on bedload can be used to extend the data set to fit equation 2 provided instantaneous discharge data are available. This is particularly important where high discharge event data are incomplete.

Application to erosion monitoring

The bedload–total sediment load relationships have application to monitoring specific parts of a mine site for sediment movement off-site, or for the design of sedimentation ponds. Using the type of erosion plots described here, intensive monitoring may only be needed in the initial years until the relationships can be established. After this period the relationships could be used to predict the amount of suspended load (generally the most mobile and potentially damaging to the environment) from the collected bedload. The power function (eqn 1) could reasonably be applied to any surface treatment on the Ranger mine landforms, provided allowance is made for under-prediction of the large runoff events.

Temporal sediment yield variations

The variation in bedload with time from the three sites is shown in figures 3B, C and D. For the fire and the soil sites the inverse log-linear relationships describing event bedload with time are significant (figs 3C & 3D). Initial bedload from the less vegetated soil site was almost twice that from the fire site which had the more complex vegetation community. Bedload from the soil site decreased with time at a greater rate than from the fire site but remained approximately constant for the unvegetated cap site. For all sites the scatter associated with the data is due to the size, duration and timing of the rainfall events.

Quadrant measurements (Hall pers comm 1996) showed a variable increase in total living ground vegetative cover from approximately 16% on 30/1/95 to 48% on 28/3/95 on the soil site and from 26% to 44% over the same time on the fire site. Living cover then decreased during dieback as the Wet season finished. The estimates of living cover are not a true estimate of total cover on the plots as there is a high percentage of leaf litter and dead vegetation present. Observations indicate this almost doubles the surface cover.

Event rainfall during the 1994/95 Wet season ranged from 7 mm to 178 mm. Rainfall events were selected to demonstrate the amount of rainfall and the associated bedload from the plots (figure 4). The three monitored plots were within 500 m of each other and it is reasonable to assume that the sites experienced the same storm events. For corresponding storms at each site, bedload was highest from the cap site. Bedload loss was higher for the soil site than for the fire site. Generally, bedload and rainfall amount are positively correlated.

There is probably a combination of two contributing factors to the reduction in bedload on the soil and fire sites: (1) a significant increase in vegetation cover during the early part of the Wet season; and (2) a possible depletion of erodible material, which was difficult to determine due to the lack of early Wet season sediment concentration data. Early Wet season rains may have eroded the sediment that might have been detached and made available during the preceding Dry season, resulting in an initial flush of sediment. Increasing vegetation cover during the Wet season coincided with a reducing bedload rate. Spear grass (*Sorghum*) were the dominant vegetation that grew during the Wet season. The dense cover of this genus contributed to sediment loss reduction.

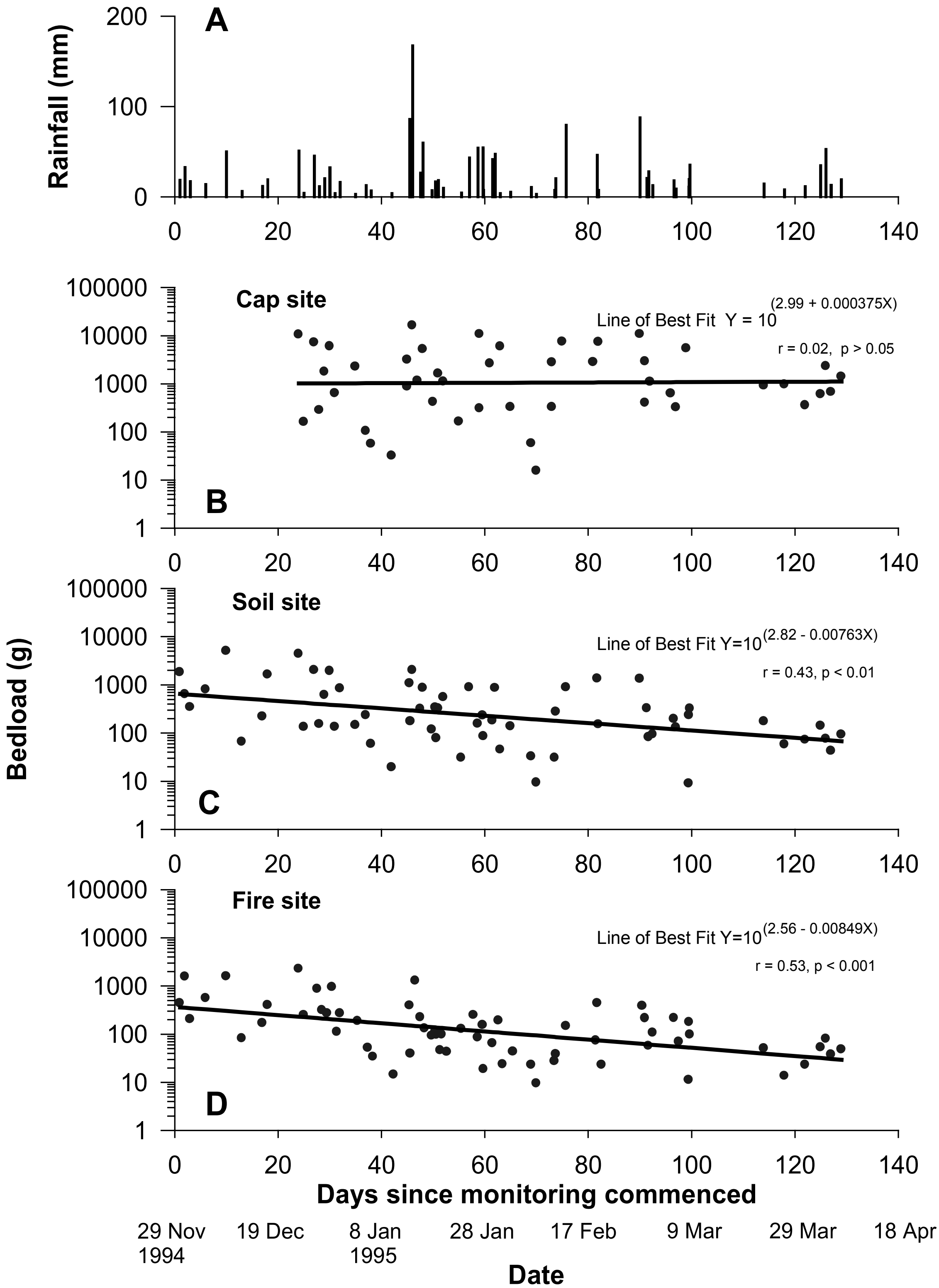

Figure 3 Bedload from the monitored sites and the corresponding rainfall

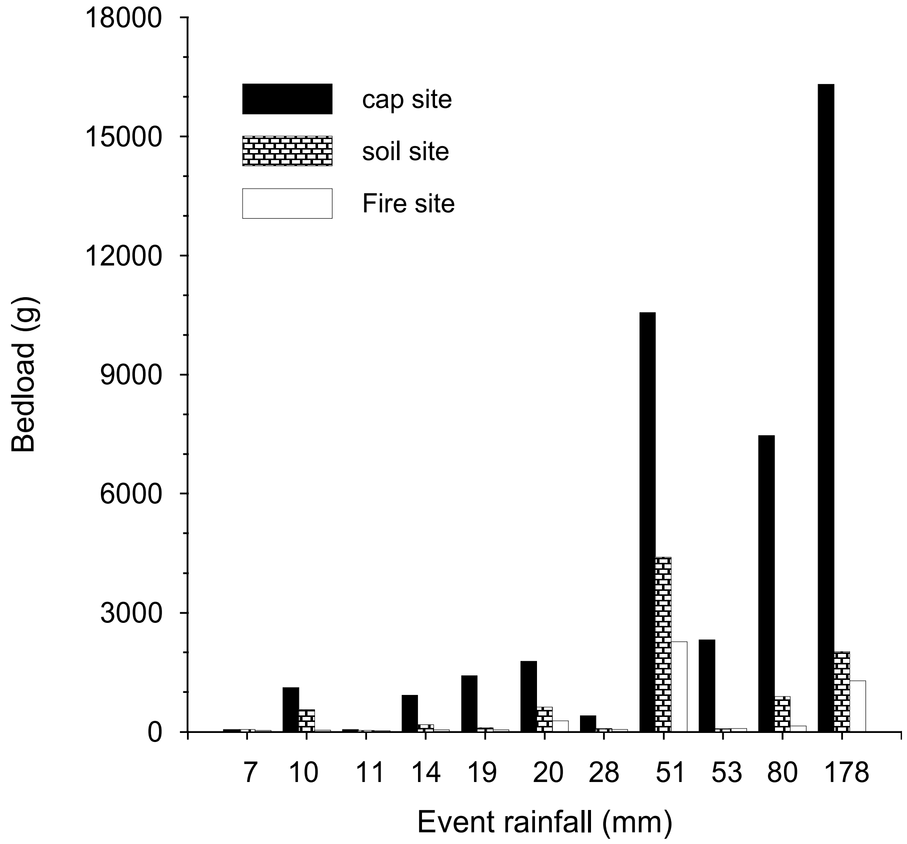

Figure 4 Bedload losses from selected rainfall events

Conclusions

The observed linear relationships between bedload and total sediment load have important implications for the derivation of modelling parameter values. In the case of the Ranger mine it is possible to collect the bedload sediment after the event and calculate the total load using the site specific sediment relationships. If sufficient storms are not observed, where suspended sediment load is measured, predicted total sediment loads could be used to calibrate landform evolution models such as SIBERIA.

The main change, occurring during the 1994/95 Wet season monitoring period, was an increase in low level vegetation cover on the soil and fire sites. Increasing vegetation cover by small shrubs and grasses (*Acacia* and *Sorghum* species) is a major factor controlling the observed reduction in sediment loss. Sediment loss from the vegetated plots reduced at a decreasing rate until reaching a practically constant rate midway through January. The results also show that the fire site with the established vegetation cover of taller trees had the least amount of bedload while the cap site with no vegetation had the greatest bedload.

Acknowledgments

The project was conducted in co-operation with ERA Ranger mine. The contributions of the following are acknowledged: Bryan Smith, Craig Bagnall, Ray Hall and Noel Powell (***eriss***); and Corinne Unger, Phil Savory, the late Dr Peter Reid, Gary Stewart, Hazel Nisbet and Richard MacAllister (ERA). Dr Wayne Erskine (State Forests of New South Wales) commented on a draft of this paper. Dr Rob Loch, Landloch Pty Ltd provided helpful discussion.

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ENVIRONMENTAL IMPACT OF MINING

Ecosystem Protection

Temporal variability of macroinvertebrate communities in Australian streams: Implications for the prediction and detection of environmental change[1]

CL Humphrey

Introduction

An accompanying paper by Humphrey et al in this volume ('AUSRIVAS operator sample processing errors') describes research carried out under the 'Monitoring River Health Initiative' (MRHI), a scheme developed and funded under the Commonwealth of Australia's 'National River Health Program'. The MRHI has involved government agencies from Australian States and Territories in developing a standardised and coordinated rapid assessment approach to biological monitoring of water quality in Australian rivers and streams. The basis of the Australian approach, AUSRIVAS (AUStralian RIVer Assessment Scheme), is the development and use of models that would predict macroinvertebrate community composition in the absence of human disturbance (Davies 2000).

It was recognised early in the development of the MRHI that temporal variability in macroinvertebrate communities in a country of climatic extremes such as Australia, could pose serious risks to development of sensitive, predictive models for biological monitoring. The issue of temporal variability of macroinvertebrate communities and possible implications to successful model development were the subject of this study.

An assumption of predictive modelling is that macroinvertebrate community composition is reasonably constant over time. This is an issue that has received only limited consideration during development of the British RIVPACS system (eg Wright 1995) on which the Australian AUSRIVAS system was based. The present broad-scale study was conducted to quantify the degree of temporal variability evident in long-term data sets from representative streams across Australia. Where lack of 'persistence' was observed, the implications of the result for model sensitivity were to be explored by assessing the degree of fidelity of long-term data in groups derived from current AUSRIVAS classifications. Some possible ways to account for temporal variability are also discussed.

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[1] More detailed discussion of this research is provided in Humphrey CL, Storey AW & L Thurtell 2000. AUSRIVAS: Operator sample processing errors and temporal variability — implications for model sensitivity. In *Assessing the biological quality of fresh waters. RIVPACS and other techniques*, eds JF Wright, DW Sutcliffe & MT Furse, Freshwater Biological Association, Ambleside, 143–163.

Methods

Measure 'persistence' of macroinvertebrate communities from long-term data sets

Data from a number of researchers across Australia were analysed to quantify the degree of temporal variability of stream macroinvertebrate communities (Humphrey et al 1997b). Ten geographical regions, 15 catchments and 38 individual sites were represented. The average duration of the data sets was approximately 6 years, with some data sets extending to 10 years. Sites were located in riffle habitat of permanent and seasonally-flowing streams.

Temporal variability was expressed in terms of an index of 'Inconstancy', determined for each site and season as the proportion of interannual comparisons of community composition (presence/absence) and structure (rank abundance) for which Bray-Curtis dissimilarity measures (family level identifications) exceeded pre-determined thresholds. As dissimilarity measures are the basis of UPGMA (Unweighted Pair Group arithMetic Averaging) classification of MRHI data for model development, these were potentially best suited to quantifying the degree of temporal variability in a data set.

Implications of lack of persistence for classification

Where lack of persistence of macroinvertebrate communities was observed in a long-term data set, an objective of the current project was to explore the implications of the results for predictive modelling by assessing the degree of temporal variability in reference sites relative to classifications of related impacted sites. Using long-term data from the upper South Alligator River (SAR) (Humphrey et al 1995a, 1997a), Humphrey et al (1995b) explored the implications of a marked switch in structure of macroinvertebrate communities (rank abundances) that occurred between pre-1993 and post-1992 time periods, by assessing whether the post-1992 data classified near or together with those from both unpolluted/mine-polluted portions of the adjacent Rockhole Mine Creek.

The ultimate test of whether or not temporal variability presents problems for predictive modelling lies in running long-term community compositional data for particular sites, such as those from the SAR, through agency classifications and models. In this context the severity of any lack of community persistence can be fully measured. Misclassifications and poor predictions would indicate potential problems for model development. To this end, the same long-term SAR data (but this time using presence–absence data) were incorporated into an NT MRHI agency classification based upon riffle samples gathered throughout the NT.

Evaluate ways to account for temporal variability and make recommendations

It became evident through the course of this study that seeking environmental correlates that may account for temporal variability would be unlikely to be successful for many of the data sets for which 'high' temporal variability was found. This is reviewed further below.

Results

Persistence of macroinvertebrate communities from long-term data sets

Degree and extent of temporal variability: rank abundance data

For half of the catchments studied, over 30% of interannual comparisons exceeded a Bray-Curtis dissimilarity value of 0.5 (table 1). Only for a relatively small portion of southern

Australia, for which interannual variability of discharge is low — Tasmania, south-west WA and possibly parts of Victoria — would there appear to be potential for development of AUSRIVAS models based upon rank abundance (community structure) data. Given this restriction and the fact that current AUSRIVAS models use presence-absense (compositional) data, the rest of the discussion focuses on results using presence–absence data.

Table 1 Temporal variability of stream macroinvertebrate communities from riffle habitat across different regions of Australia, based upon family-level, presence–absence (PA) or relative abundance (RA) data. Inconstancy indices are averaged across seasons.

Region	Flow status (Permanent or Seasonal)	Inconstancy index, PA (%dissim >0.35)	Inconstancy index, RA (%dissim >0.5)	CV of annual flow	Latitude (degrees & minutes)
Temperate (VIC-Latrobe)	P	4.5	68.0[1]	0.32	38.0
Temperate (SW WA)	P	6.5	3.5	0.49-0.73	32.3
Temperate (TAS)	P	7.5	5.5	0.47	41.3
Wet-dry tropical (SAR, NT)	P	13.5	40.5	0.58	13.35
Wet tropical (NE QLD)	P	15.0	33.5	0.5	18.1
Wet-dry tropical (RMC, NT)	S	17.0	0	0.58	13.35
Subtropical (SE QLD)	P	19.0	52.5	1.04-1.07	26.3
Temperate-dry (VIC-Wimmera)	S	19.5	41.5	0.58-0.98	36.3
Wet-dry tropical (Magela, NT)	S	21.0	36.0	0.56	12.4
Temperate (SW WA)	S	24.0	15.0	0.49-0.73	32.3
Temp. semi-arid (Flinders, SA)	P (riffle)	25.0	6.0	1.25	31.1
Sub-alpine (NSW)	P	27.5	37.5	0.5-0.75	36.3
Temp. semi-arid (Flinders, SA)	P (MH[2], pool)	51.7	16.5	1.25	31.1
Dry tropics (NW, WA)	S	93.0	27.0	1.4	21.3

[1] Data gathered using a sample processing method inappropriate for recovering relative abundance data; [2]MH = macrophyte habitat.

Degree and extent of temporal variability: presence–absence data

A combined-seasons index was derived by averaging the Inconstancy index across seasons for presence–absence data (table 1). Regression analysis was used to seek relationships between dependant Inconstancy index and independent environmental variables. The best predictive relationship was one derived between the Inconstancy index variable and the independent variables, Coefficient of Variation of annual flow and flow status (permanent/seasonally-flowing) ($R^2 = 0.77$).

Three summary points may be made from the results of Humphrey et al (1997b) and from regression analysis. (The term ‘persistence’, the converse of ‘inconstancy’, is used to describe the degree of similarity in community composition over time.)

1a A high negative correlation is observed between persistence and interannual variation of stream discharge.

1b Persistence of macroinvertebrate communities is significantly higher in streams of permanent flow than in streams of seasonal flow.

2 Measures of temporal variability used in the study indicated relatively high persistence of macroinvertebrate communities for all but one or two regions represented. For regions exhibiting high Inconstancy index values, cyclonic disturbance and flooding were attributed as the cause (Humphrey et al 1997b). Nevertheless, the ‘high temporal variability’ regions represent a large portion of the continent. In particular, Humphrey et al (1997b) extrapolated the findings to suggest that the sensitivity of AUSRIVAS models developed for much of the drought-prone portion of eastern Australia, particularly NSW and QLD, could be compromised during (and possibly after) drought.

Implications of lack of persistence for classification

From ordinations that were conducted using SAR data, post-1992/pre-1993, and data from both unpolluted/mine-polluted portions of the adjacent Rockhole Mine Creek (RMC), Humphrey et al (1995b) showed that the magnitude of change occurring in the SAR post-1992 was even more severe than that occurring in polluted portions of RMC. Moreover, the direction of change occurring in the SAR data was in the same direction as the pollution gradient in RMC.

The limitations of this approach to MRHI modelling, however, are twofold: Firstly, the analysis for SAR-RMC was based upon family-level abundance data. Had the analysis been repeated using presence–absence data, it would probably indicate little change in SAR community composition between post-1992 and pre-1993 relative to that between the two RMC sites. Secondly and as described above, the better test of whether or not temporal variability presents problems for predictive modelling lies in running long-term data for particular sites, such as those from the SAR, through agency classifications and models.

Humphrey and Doig (1997) describe results of a classification incorporating long-term SAR data into an NT MRHI agency classification. Results showed misclassification of early (1988) SAR data in a UPGMA classification based upon late Dry season 1994 and 1995 NT riffle data, while for successive years of data (1994 and 1995), about 50% of the 15 comparable NT sites occurred in different classification groups. However, because of the low interannual pairwise dissimilarity, low inter-site dissimilarity generally, and the fact that the classification was based on few sites (less than 25), no firm conclusions could be drawn from the study. Consequently, the full implications of any lack of temporal variability present in other long-term data from elsewhere for agency model development, accuracy and precision, will require similar approaches to that used for NT data.

Evaluate ways to account for temporal variability and make recommendations

Where 'significant' temporal variability is found after running long-term data for particular sites through agency classifications and models (viz misclassifications and poor predictions), possible approaches to dealing with this variability for predictive modelling include:

1 Risk-based assessment using AUSRIVAS models

Predictive regression relationships between temporal variability and environmental variables as described above, may be used to quantify degrees of 'risk' of model failure for a particular location. This would give managers some indication of how useful and accurate models might be that are developed for a particular location, ie what degree of error could be associated with predictions if temporal variability was the sole factor of concern.

2 Accounting for temporal variability

As an improvement upon (1), can temporal variability be accounted for?

(i) Modelling temporal variability

Humphrey et al (1997b) concluded that seeking environmental correlates that may account for temporal variability would be unlikely to be successful for a number of situations: (i) seasonally-flowing streams where shifts in community composition over time may be associated with stochastic recolonisation processes (see also Wright 1995); (ii) longer-term (several years) recovery and recolonisation of streams following massive disturbance (eg Robe R, north-west WA); and (iii) switches between different community 'steady states' where triggers for the switch may be clearly identified, but the trajectory of community composition thereafter is either lagged, or unknown and unpredictable (eg SAR and Yuccabine Ck, north-east QLD). Associated with these difficulties is the possibility of inter-catchment differences in community responses, as described for the South Alligator R and nearby Magela Ck (NT) in Humphrey and Doig (1997).

Modelling of drought-related changes to macroinvertebrate communities would be particularly useful for AUSRIVAS model development in eastern Australia. However, there is presently little understanding of the responses of macroinvertebrate communities to drought. Moreover, Humphrey et al (1997) reported very different responses to drought across Australia at regional and inter- and intra-catchment scales. Examination of existing agency data sets, some of which span periods of major drought (eg QLD, 1994–1995) would assist in redressing these information deficiencies.

(ii) Adjusting and updating model output

This would entail the re-sampling of suitable reference sites simultaneously with monitoring sites in order to adjust model output by some factor. A problem with this approach is that it assumes the 'correction' factor is similar across classification groups and between reference and disturbed sites. This assumption is unlikely to hold because, as described in (i) above, macroinvertebrate response to a similar disturbance may differ at different spatial scales. In addition, the degree of change to natural disturbance is likely to be greater for reference sites than for human-disturbed sites. Hence sufficient reference sites would have to be included that were representative of each of the classification groups, as well as re-sampling of selected disturbed sites, in order to derive appropriate scaling factors.

(iii) Models for different climatic conditions (especially drought vs non-drought)

Models empirically derived for different climatic conditions, such as drought vs non-drought, would have the advantage that fewer assumptions are made about the responses of macroinvertebrates in different habitats, between different parts of a catchment, amongst catchments, or across a disturbance gradient. However, this approach would be expensive and there is also the untested assumption that responses to one drought will be the same as the next, even though droughts differ in their intensity. At best, interpolation and extrapolation between different models may enable some allowance to be made for different climatic conditions. Nevertheless, some of the current agency data sets span a period of 'drought' and 'non-drought'; processing of all these data and derivation of different models for different climatic conditions may be valuable.

(iv) Combined-seasons/years models

Temporal variability can be reduced substantially where data for different seasons of the year or consecutive years for the same season are combined. One disadvantage with this approach is the need to accumulate two seasons/years of data before an assessment of water quality based upon macroinvertebrate communities can be made. This may provide some indication of longer-term severity of a water quality problem but it is contrary to the ethos of rapid biological assessment and rapid turn-around of results. Moreover, it may result in construction of a model so robust and overly-inured to natural environmental change that only impacts of a particularly severe nature are detected while impacts isolated to only one of the seasons may pass undetected.

Related to approaches (iii) and (iv), some agencies have constructed models by adding new reference sites gathered for a given season and from consecutive years of sampling, to an existing model (eg UK RIVPACS, MRHI ACT agency). Without simultaneous sampling of some common reference sites to account for possible temporal variation, this approach runs the risk of deriving models that are temporally confounded.

Some combination of approaches (ii) and (iii) may provide adequate solutions to developing AUSRIVAS models that account for temporal variability. For some geographical regions, temporal variability may be too large for useful predictive models to be developed. For these situations, it may be necessary to resort to more traditional hypothesis-testing approaches involving BACI designs and derivatives (eg Underwood 1991, Faith et al 1995).

Ongoing research

While some preliminary data simulations have been undertaken to determine the consequences to model development and sensitivity of temporal variability, a more complete sensitivity analysis is currently underway under Phase II of the National River Health Program to determine the full implications of variability (at various spatial scales) for model sensitivity. This analysis includes quantifying variation, as well as other error sources, and their effects on the rates of misclassification to quality bands (*sensu* Clarke et al 1996).

It would be prudent to be cautious in the promotion of AUSRIVAS for site-specific assessments until the sensitivity of the AUSRIVAS methodology has been fully assessed and data quantity increased and quality improved.

Acknowledgments

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An overview of the new water quality guidelines for Australian and New Zealand aquatic ecosystems[1]

CL Humphrey & KW McAlpine[2]

Background

The Australian and New Zealand Environment and Conservation Council (ANZECC) and the Agriculture and Resource Management Council of Australia and New Zealand (ARMCANZ) developed the National Water Quality Management Strategy (NWQMS) to provide a consistent framework for the sustainable use of water resources across Australia and to assist all levels of government and the community to manage these resources together (ANZECC & ARMCANZ 1994). A major focus of the NWQMS is to protect and maintain different (and sometimes competing) water resource values. The approaches for achieving these objectives are an important component of the *Australian Water Quality Guidelines for Fresh and Marine Waters* released in 1992 (ANZECC 1992).

In 1993, the ANZECC Standing Committee on Environmental Protection agreed that the Guidelines should be revised in due course to incorporate current scientific national and international information. Increasing scientific understanding of the complexity of ecosystems and food production systems has meant that new ways of managing water quality are replacing the more traditional scientific and management approaches. These more holistic, best practice approaches are designed to ensure that water resources are managed sustainably.

The Environmental Research Institute of the Supervising Scientist began the task of coordinating the revision of the Guidelines in 1996. The scope of the revision was extended so that the new Guidelines would also relate to water resources in New Zealand. The review process involved public consultation, establishment of technical groups to cover different subjects and phases of the revision process and ANZECC/ARMCANZ agency involvement to ensure the approaches and methodologies for the different sections were up-to-date, consistent and compatible with current policy at all levels of Government.

Philosophical basis

The new Guidelines have an expanded or new focus in important areas:

Holistic management. It is now well recognised that all aspects of the environment are interdependent and that influences on the environment can not be considered in isolation. For example, clearing of river bank vegetation combined with elevated nutrients from rural or urban catchments can give rise to algal blooms. The different environmental values (or uses) are also interdependent and must be considered together. Thus, types of ecosystems, food

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[1] This paper is summarised from McAlpine K & Humphrey CL 2001. An overview of the guidelines for Australian and New Zealand aquatic ecosystems. *Australasian Journal of Ecotoxicology* 7, 41–50.

[2] Department of Environmental Protection, PO Box K822, Perth WA, 6842.

production systems, interactions, cumulative effects and modifying factors must all be examined in studying water quality.

Cooperative best management. Environmental management in Australia and New Zealand is moving towards a 'cooperative best management' approach where the emphasis is on prevention, environmental outcomes and cooperation rather than control, prescriptive regulation and direction. This contemporary approach involves industry, Government and community working together to meet agreed environmental outcomes.

Water quality vs environmental quality. For many aquatic ecosystems, factors other than water (or sediment) quality may limit or prevent the achievement of management goals. It may be more appropriate to spend the majority of resources on programs such as habitat restoration or control of exotic pest species, ahead of water quality issues. The Guidelines should be integrated into, and implemented as part of, collective environmental guidelines (eg water quality, environmental flows and riparian condition).

Focus on managing issues. To protect designated environmental values the new Guidelines promote the use of management goals based on the environmental issues of concern, rather than the concentration of individual contaminants. By identifying the issues (eg algal blooms, deoxygenation, toxicity) the appropriate water quality indicators can be selected together with the environmental processes that can directly or indirectly affect the indicator (eg light, flow and algal grazing can affect chlorophyll *a* concentration). It is then possible to identify and take into account secondary, site-specific factors that can alter the effect of the threatening contaminant, or the severity of its effect. A guideline can then be applied according to the environmental/biological conditions prevailing at a specific site. This is an iterative risk-based process that leads to an improved estimate of the risk of an impact occurring. One of the key advances made in the revised Guidelines is the provision of risk-based decision frameworks to assist the user to make these site-specific assessments.

Continual improvement. Continual improvement is a fundamental principle embraced in water quality management strategies in Australia and New Zealand. Managers are encouraged to promote 'best practice' and improve the quality of water resources — using intermediate quality targets if necessary — rather than allowing them to degrade.

Integrated assessment. Currently, scientific understanding of the environment is insufficient to allow confident predictions about how a particular concentration of a contaminant will affect an ecosystem. Therefore, biological as well as physical and chemical aspects must be measured to confidently assess whether contaminants have significantly affected ecosystem health. This integrated approach also acknowledges assessment of other key environmental indicators apart from water quality (see above). Integration also needs to be across the whole catchment to ensure management considers cumulative impacts and impacts on environmental values in downstream ecosystems, such as estuaries or coastal waters.

Implementation

Water quality management involves the consistent, and preferably integrated, use of the range of legislative and regulatory tools at the national, state/territory and regional or catchment levels, as well as community action. However, society must have a collective vision of what it wants for each water resource and there needs to be a good technical understanding of human impacts and their control. After the available technical information for a specific water resource has been collated, the steps described below in the water quality management framework could provide a consistent national approach to managing water quality.

The management framework

The management framework is based on the premise that overall responsibility for water resource management rests with the community. The tools, strategies and policies developed to manage and protect environmental values are applied in this wider context. With this in mind, a water quality management framework has been developed for managing water quality consistently across Australia and New Zealand (fig 1). At each step in the framework the community, government and industry are encouraged to work together cooperatively so that management strategies can be developed and implemented effectively.

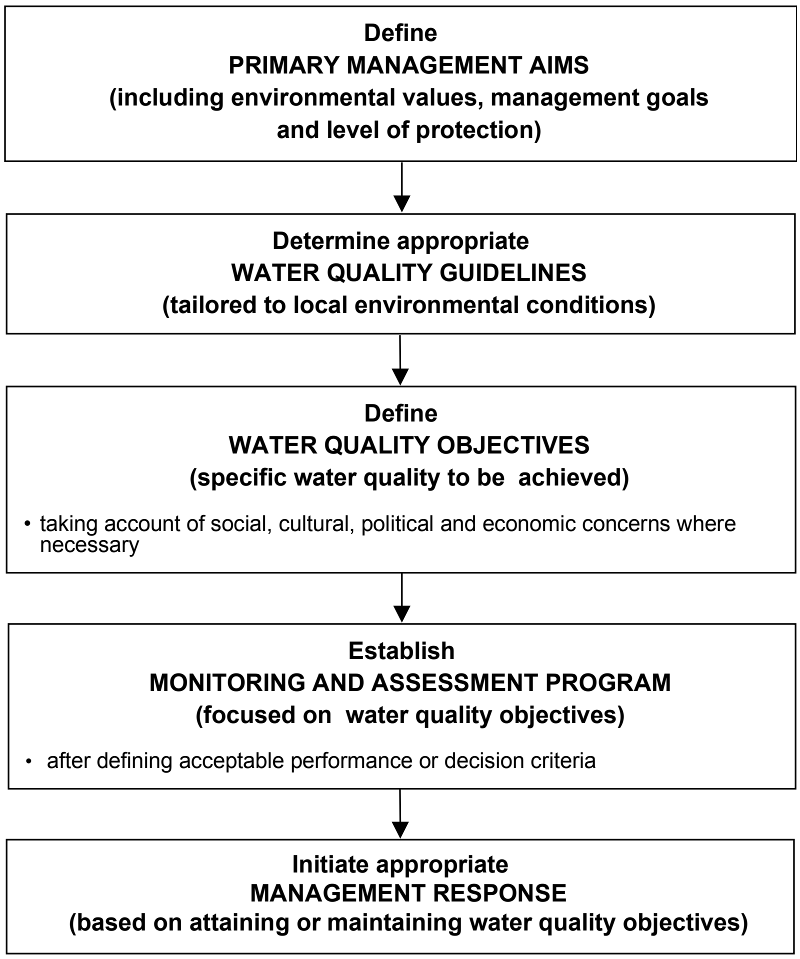

Figure 1 Management framework for applying the water quality guidelines (from ANZECC & ARMCANZ 2000a)

The first step is to identify the *environmental values* for a water resource through community involvement. Environmental values are particular values or uses of the environment that are important for a healthy ecosystem or for public benefit, welfare, safety or health and which require protection from the effects of pollution and waste discharges (ANZECC & ARMCANZ 2000a). Six environmental values are recognised in the Guidelines:

- aquatic ecosystems,
- primary industries (irrigation and general water uses, stock drinking water, aquaculture and human consumers of aquatic foods),
- recreation and aesthetics,

- drinking water,
- industrial water (no water quality guidelines provided), and
- cultural and spiritual values (no water quality guidelines provided).

In most cases more than one environmental value would apply to a water resource and managers would need to achieve the water quality of the most conservative of the values.

A primary focus for the new Water Quality Guidelines, and indeed the greatest interest and use of past guidelines, has been on the management of water resources to protect *aquatic ecosystems*. An outline of the important advances in the Guidelines for protection of aquatic ecosystems is the primary focus of the following discussion. McAlpine and Humphrey (2001) provide more detailed information about the Guidelines.[1]

For each environmental value *management goals* need to be formulated, in consultation with stakeholders, describing more precisely what is to be protected. Management goals need to be achievable, measurable and reflect community needs and desires. Examples could be to reduce the occurrence of algal blooms, or provide water quality safe for swimming.

A *water quality guideline* is a numerical concentration limit or descriptive statement recommended to support and maintain a designated environmental value. To protect aquatic ecosystems, guidelines are provided for four broad indicator types, ie:

1. *Biological indicators*. For example algae, macrophytes, macroinvertebrates and fish;
2. *Physical and chemical stressors.* These are natural water quality parameters: nutrients; biodegradable organic matter; dissolved oxygen; turbidity; suspended particulate matter; temperature; salinity; pH and changes in flow regime;
3. *Toxicants in water*. 'Toxicants', the term given to chemical contaminants such as metals, hydrocarbons and pesticides that can potentially have toxic effects at concentrations that might be encountered in the environment; and
4. *Sediment toxicants.* Sediments are a sink for many contaminants that can adversely affect benthic and other aquatic organisms.

A summary of the development, rationale and application of guidelines for each of these broad indicator types is provided in two special issues of the *Australasian Journal of Ecotoxicology*, volume 7.

For the physical and chemical (non-biological) indicators, the guidelines are termed 'trigger values' because if monitoring (or test site) data exceed the value, a management response is triggered — either conduct further site specific investigations to assess whether or not a problem really exists (using the risk-based decision frameworks) or instigate remedial action.

The preferred approach to deriving the guideline trigger values is using local effects-based data or, for some indicators and circumstances, using data from local reference sites. In the absence of these data, default trigger values are provided using regional reference data (physical and chemical stressors) or global biological effects (toxicological) databases (toxicants and sediments). In some situations, guideline trigger values may be modified to site-specific guidelines using the decision frameworks described later in this paper.

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[1] The web site for Australian water quality guidelines for fresh and marine waters is www.ea.gov.au/water/quality/nwqms.

Water quality objectives are the specific water quality targets agreed between stakeholders, or set by local jurisdictions, that are used to report on the performance of management strategies in meeting the management goals. They are based on water quality guidelines but may be modified by other inputs such as social, cultural economic or political constraints.

The established water quality objectives would generally be incorporated into water quality management plans, programs and strategies (including the use of regulatory instruments) that aim to protect the designated environmental values.

Monitoring and assessment programs are an essential component for measuring environmental performance and checking that the management goals are being achieved and the environmental values protected. The Water Quality Guidelines and their companion document, the *Australian Guidelines for Water Quality Monitoring and Reporting* (the Monitoring Guidelines) (ANZECC & ARMCANZ 2000b) contain information on the practicalities of designing monitoring programs and collecting and analysing data for the measurement of water quality. The Monitoring Guidelines contain a framework that sets out basic, general steps and details of how to plan a monitoring program, while the Water Quality Guidelines contain more detailed information specific to issues raised in its chapters.

New approaches to the protection of aquatic ecosystems

Although natural variations in the physical and chemical attributes of ecosystems can have important consequences for aquatic ecosystems, human-induced changes can be far more profound, and in many cases they can be effectively irreversible. The objective adopted for the protection of aquatic ecosystems is therefore: '*to maintain and enhance the 'ecological integrity' of freshwater and marine ecosystems, including biological diversity, relative abundance and ecological processes*' (ANZECC & ARMCANZ 2000a).

Depending on whether the ecosystem is non-degraded or has a history of degradation, the management focus can vary from simple maintenance of present water quality to improvement in water quality so that the condition of the ecosystem is more natural and ecological integrity is enhanced.

A more holistic set of environmental quality guidelines

Aquatic ecosystems cannot be considered as static environments — their biota, physical structure and chemistry all fluctuate according to seasonal and climatic influences and according to catchment vegetation and landuses. Water, sediment and biota are all in intimate contact with each other and the partitioning of chemical contaminants between these three media is under constant flux. Measurement of the biological components indicates whether in fact the ecosystem has been adversely affected by human activities and measurement of the physical and chemical indicators gives some insight into the cause of an observed change in the biota, or can be used to give early warning of potential impacts on the biota. To this end, the Guidelines provide guidelines for, and promote monitoring of, both biological and chemical components of surface waters and sediments.

Three levels of protection

It is unrealistic to impose uniform management goals, guidelines and regulatory frameworks across the spectrum of ecosystem conditions. To assist users, different sets of guidelines are recommended for each of three levels of protection, based on ecosystem condition:

- high conservation/ecological value systems;
- slightly to moderately disturbed systems (where the guidelines will mostly apply); and

- highly disturbed systems.

A management goal of ‘no change’ to biological diversity is recommended for systems of high conservation or ecological value or quality, and local biological effects and monitoring data are of overriding importance in guideline trigger value derivation and assessment of test data. Until such biological data are available, the recommended starting point for all indicators is generally no change beyond natural variability for this level of protection.

The default trigger values provided in the Guidelines are generally intended for use in slightly to moderately disturbed systems. If these values are unsuitable for highly disturbed systems, more relaxed (less protective) defaults are available for toxicants in water. Nevertheless, for both of these ecosystem conditions, local reference data (see following section) or preferably, local biological effects and monitoring data, will assume greater importance in trigger value derivation and test site assessment.

Accounting for variability between and within different ecosystem types

It is unrealistic to expect a uniform set of guidelines to apply across all ecosystem types and all regions and, therefore, effective management must incorporate site-specific information.

Firstly, it involves classifying the water resource according to *ecosystem type*. Guidelines have been tailored as far as possible to different broad ecosystem types.

Secondly, a greater emphasis has been placed on the use of reference sites to define a *reference condition* that can be used to provide the quality targets for management to achieve, as well as meaningful comparisons to use in monitoring and assessment programs.

Thirdly, an appropriate *level of protection* must be selected based on the target condition for the ecosystem.

Finally, *risk-based decision frameworks* are provided to help address the issues of ecosystem variability and complexity, giving a more realistic estimate of the ecological risk arising from contamination of the environment. They help the user to refine the measured test value so that it is appropriately compared with the conservative single-number guideline trigger value, for local application. In addition, the frameworks may be used to refine the trigger values themselves. In either case, they allow the user to take into account local environmental factors that may potentially affect the action of a particular contaminant, bringing the values closer to those that may elicit a biological response. For biological indicators, a different decision framework is incorporated since the measured biological responses are actually the management end-points in the decision trees for the physical and chemical indicators.

A generalised example of the risk-based decision tree is shown in figure 2. In most cases use of the decision trees will only be triggered once the level of a guideline trigger value has been exceeded. Through the decision frameworks, additional site-specific information is obtained on a step-by-step basis to modify the test site values and re-assess whether the guideline is exceeded or not. While some simple trigger value refinements can be achieved upfront, for most indicators and issues, trigger values are refined using the decision trees only after continuous and extensive monitoring shows that test site data exceedances are consistently assessed as posing no risk to the ecosystem. For whichever application the decision trees are used, the initial steps are relatively simple adjustments, but the ultimate steps are more resource intensive, usually requiring local toxicological data to be gathered.

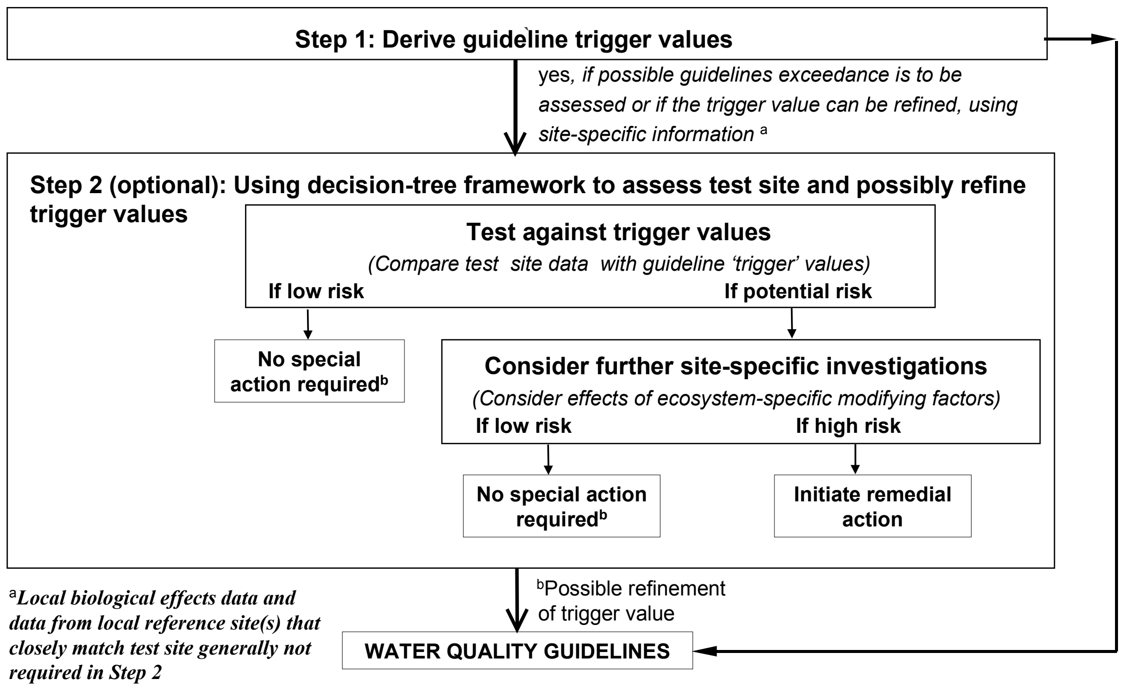

Figure 2 Use of decision trees for assessing test sites and refining trigger values, for physical and chemical indicators in water and sediment. Adapted from ANZECC and ARMCANZ (2000a).

Use of the decision frameworks is not mandatory, but they can reduce the amount of conservatism built into the guideline trigger values allowing more realistic targets to be achieved. If managers do not use the decision frameworks then their alternative option is to apply the trigger values as default guidelines. Obviously there is a cost associated with using the frameworks just as there is a potential (remedial) cost associated with not using them for cases where a test value exceeds the trigger value. All stakeholders will need to be involved in balancing the costs and the benefits of each option. Ultimately, the availability of data, expertise, resources and time will determine which, if any, steps in the framework are used.

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AUSRIVAS operator sample processing errors — implications for model sensitivity[1]

CL Humphrey, AW Storey[2] & L Thurtell[3]

Introduction

In 1993 the Commonwealth of Australia funded the 'National River Health Program' (NRHP) to monitor and assess the health of the nation's rivers and streams (Schofield & Davies 1996). Part of this program was the 'Monitoring River Health Initiative' (MRHI), involving government agencies from all Australian States and Territories to develop a standardised and coordinated rapid assessment approach to biological monitoring of water quality in Australian rivers and streams.

The River InVertebrate Prediction And Classification System (RIVPACS; Wright 1995) was adopted as a national framework for the Australian program. Models were to be based on family-level identifications of macroinvertebrates collected by habitat-specific kick-sweep sampling at an initial 1400 reference sites sampled across Australia in two seasons (Schofield & Davies 1996). A full description of the program as developed to 1997, AUSRIVAS (AUStralian RIVer Assessment Scheme), is provided by Davies (2000).

eriss was contracted by the NRHP to develop and implement programs for Quality Assurance/Quality Control (QA/QC) of key aspects of the MRHI protocol, including agency sample processing procedures. While AUSRIVAS has adopted a standardised approach for sample collection, agencies opted for one of two approaches for sample processing in Phase I of the program (1993–96): 30 min live-sort of each sample in the field (QLD, NSW, VIC, TAS & south-west WA) or field preservation of samples and subsequent laboratory-based subsampling and sorting (NT, SA, ACT and north-west WA). It was recognised early in the development of the MRHI that sample processing error due to live-sorting of samples in the field could pose serious risks to development of sensitive, predictive models for biological monitoring. To this end, the nature and degree of error arising in agency sample processing procedures was quantified and the implications of high error rates for model sensitivity explored by way of preliminary simulations. The results of this investigation are summarised in this paper.

Methods

QA/QC procedures for assessment of agency sample processing performance

Sample residues left after agency sample processing (field live sorting and laboratory subsampling and sorting) were selected at random and transported to ***eriss*** for external

[1] More detailed discussion of this research is provided in Humphrey CL, Storey AW & L Thurtell 2000. AUSRIVAS: Operator sample processing errors and temporal variability — implications for model sensitivity. In *Assessing the biological quality of fresh waters. RIVPACS and other techniques*. eds JF Wright, DW Sutcliffe & MT Furse, Freshwater Biological Association, Ambleside, 143–163.

[2] Department of Zoology, The University of Western Australia, Crawley, WA 6009.

[3] Formerly ***eriss***. Current address: NSW Department Land and Water Conservation, PO Box 136, Forbes, NSW 2871.

QA/QC processing. Processing entailed subsampling and sorting of residues and comparison of macroinvertebrate community composition and structure data present in an estimate of the ‘whole sample’ (live-sort + residue) with those present in the agency component. Agency data were assessed against the degree of departure in taxa number and community composition from whole sample estimates (WSE). Complete descriptions of this work are contained in Humphrey and Thurtell (1997).

Implications of errors associated with live sorting for modelling

As reported below, errors arising from live sorting were associated with biases in taxa recovery such that small and cryptic taxa commonly occurring in samples and across sites were often missed. A simulation study was undertaken to address the consequences of missed common taxa to model construction and performance. Sample processing errors were introduced into an agency UPGMA (Unweighted Pair Group arithMetic Averaging) classification and model that was relatively ‘error-free’ (ie derived from a laboratory subsampled and sorted data set).

Specifically, the taxa in an AUSRIVAS data set for which data on taxa commonly-occurring across sites were well represented (ie an ACT subsampled and sorted data set) were altered to match the bias observed in live-sort data. Two sets of live-sort data were used in the simulations: NSW, one of the poorer performing agencies, and for the average bias observed across eastern states, QLD, NSW, VIC and TAS. The average bias was not as severe as that for the single agency. Deletion of taxa was performed at random from actual occurrences in the original ACT data set, until the occurrences matched that of the bias represented in the two data sets. The deletions involved 16 out of a total of 39 taxa. For each of the single agency and average agency data, 10 separate simulations and classifications were run.

Results

Agency sample processing performance

Humphrey and Thurtell (1997) describe results of the external QA/QC audits. For agencies using a live-sort method for sample processing, two main sources of error were identified from assessment of 95 agency samples, ie (i) under-representation of taxa; and (ii) different taxa recovery rates depending upon the efficiency of the operator. Factors contributing to poor taxa recovery in live-sorted samples included (i) low live-sort sample size, (ii) operator inexperience (see figure 1), and (iii) taxa commonly occurring in samples and across sites being missed (Humphrey & Thurtell 1997).

For agencies using a lab subsampling and sorting method for sample processing, the main errors associated with the 40 samples examined were poor taxa recovery at low sample size, a consequence mainly of proportional subsampling (Humphrey & Thurtell 1997).

For MRHI, recovery of taxa that have a frequency of occurrence in a group of >50% is particularly important as these taxa are used in modelling and represent taxa expected at a site. To quantify the extent to which these key taxa were being missed from live-sort samples, taxa occurring in more than 50% of samples from any of the agencies, for either the live-sort or corresponding WSE component were listed. For each of these taxa and for each agency, the percentage occurrence amongst all samples for which the taxon was found in both LS and corresponding WSE components was recorded (figure 2).

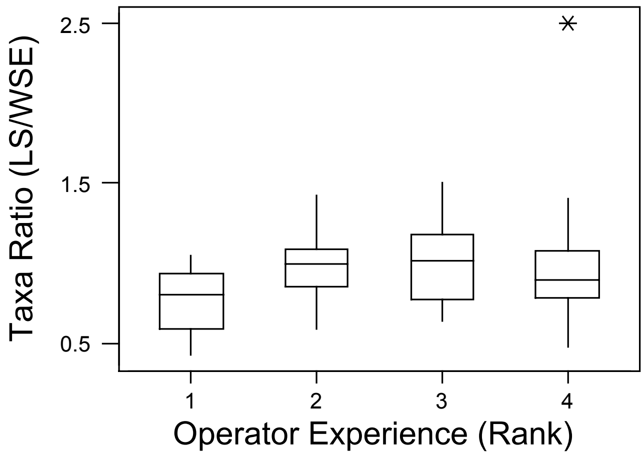

Figure 1 Boxplot showing comparison of LS-WSE (presence–absence data) for operators of different levels of live-sorting experience using the LS/WSE ratios of taxa number. Boxplot defines lower and upper quartiles divided at the median, with vertical lines showing the range of values that fall within 1.5 times the interquartile range. Outlier (point outside these limits) plotted with asterisk. Levels of operator experience for Ranks: 1 = < 2 years, 2= 2–3 years, 3 = 3–4 years, 4 = 5 years or more.

In figure 2, taxa have been ranked, from left to right, from greatest deficit to greatest surplus in occurrence in live-sort component compared to occurrence in corresponding WSE, when data were averaged across all live-sort agencies. Comparisons between occurrences of taxa present in the live-sort and corresponding WSE components show that similar taxa were either missed or better represented (in comparison to WSE occurrence) across all agencies and operators. Thus, there is consistency amongst all agencies in the biases in taxa recovery.

The results show that small and/or cryptic taxa are often overlooked during the live-sort process, regardless of agency. Thus, chironomid pupae and other small cryptic Diptera such as ceratopogonids and empidids were frequently missed during live-sorting (figure 2). This is also the case with the cryptic elmid larvae, hydroptilids (micro-caddis) and oligochaetes.

Figure 2 also demonstrates that some taxa are better represented in live-sort data than WSE (= laboratory subsampled and sorted) data. Thus the large but less abundant taxa, such as the odonates, shrimps and adult beetles, are often missed during the more objective subsampling process. These results show the extent to which laboratory subsampling and sorting are biased in taxa recovery. In practice, this is minimised for most MRHI agencies which process samples in the laboratory because an additional search of the entire sample for large taxa missed during the subsampling process is usually carried out.

Implications of errors associated with live sorting for modelling

The rate of bias against common taxa due to live sorting (single agency and average of 4 agencies) was superimposed upon an AUSRIVAS dataset in which common taxa were well represented (ACT laboratory subsample agency). The consequences of missed common taxa for UPGMA classification were examined.

In the original ACT classification, six clearly-defined groups were identified and a predictive model was successfully constructed. However, in the classifications derived after error rates were applied to the ACT data, there was little evidence of preservation of group structure. All classifications exhibited dilation and breakdown or ‘chaining’ in classification structure.

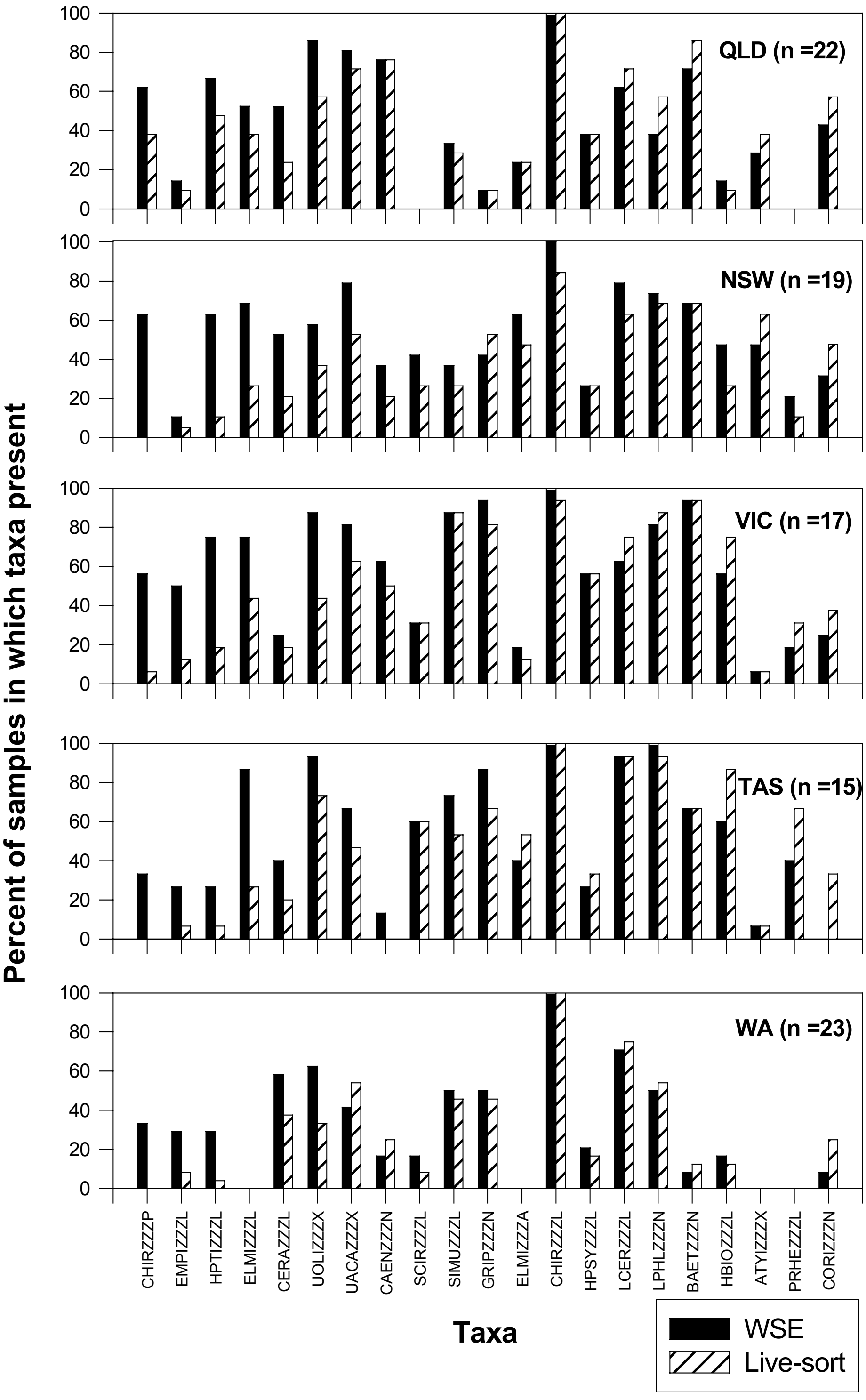

Figure 2 Taxa occurring commonly across samples for different MRHI agencies and their percentage occurrence (presence–absence data) in both LS and corresponding WSE components of agency samples. N = number of samples examined per state.

Key to taxon codes

ATYIZZZX, Atyidae; BAETZZZN, Baetidae; CAENZZZN, Caenidae; CERAZZZL, Ceratopogonidae; CHIRZZZL, Chironomidae; CHIRZZZP, Chironomidae; CORIZZZN, Corixidae; ELMIZZZL, Elmidae; ELMIZZZA, Elmidae; EMPIZZZL, Empididae; GRIPZZZN, Gripopterygidae; HBIOZZZL, Hydrobiosidae; HPSYZZZL, Hydropsychidae; HPTIZZZL, Hydroptilidae; LCERZZZL, Leptoceridae; LPHLZZZN, Leptophlebiidae; PRHEZZZL, Philorheithridae; SCIRZZZL, Scirtidae; SIMUZZZL, Simuliidae; UACAZZZX, Unidentified Acarina; UOLIZZZX, Unidentified Oligochaeta.

Suffixes L, N, X, A, P on codes refer to life stages Larvae, Nymph, Life stage not identified, Adult and Pupae respectively.

There was no evidence that classifications based on average agency (with lower error rates) were an improvement on single agency classifications (higher error rates). Even for one or two (single agency) classifications for which chaining in classification was least evident, there was a loss of two or three groups, whilst dissimilarity cut-offs for the groups were found to be higher in the altered data, indicating introduction of errors (ie higher mean within-group dissimilarities).

Placement of the simulated data site by site through an ACT model constructed using the original ACT data resulted in a high incidence of predictive failure (Humphrey et al 2000). When original data were run through the model, the majority of sites were recorded as Reference (Band A) — as expected. However, approximately 80% of sites from simulated data were recorded as impacted when run through the model, compared with 10% from the original data.

Part of the general failure to derive well-defined classifications in both sets of simulations may be related to the level of taxonomic resolution used for MRHI. Family-level presence–absence data may be so coarse that any structure present in the classifications may be easily lost. The effect of errors due to differences in sorting efficacy will depend to a large extent on the magnitude of the real differences in the data set being analysed; error will have a greater effect in a data set with small differences (short gradients). This is likely the case for the ACT, where group definition based upon family-level presence–absence data could be expected to be quite subtle because of minimal biogeographical signal. All other agency classifications are derived for the entire state or territory, minimising this problem.

Consequences of this study for ongoing AUSRIVAS programs

As a consequence of the findings reported above, aspects of the live-sort protocol were revised for implementation during the First National Assessment of River Health in Australia (1996–97). The 30 minute time limit for sorting was replaced by a target sample size of 200 animals or sorting to one hour, whichever was reached first, while agency staff were made aware of the taxa commonly missed in samples so that training programs could be implemented to redress deficiencies. In addition, poorly-sampled sites (identified by low sample size) were re-sampled to replace reference site data of dubious quality. Additional changes to the protocol will follow as the results of further R&D appear. For laboratory subsampling agencies, some recommended changes leading to standardisation of protocols have been made. These include (i) an emphasis on maximising taxa recovery (including 'large rares') through a coarse-screen search of the entire sample, and (ii) fixed-count subsampling.

Internal and external QA/QC must accompany all future sampling and sample processing by MRHI agencies so that unacceptable quality can be detected promptly to allow remedial action.

A more complete sensitivity analysis is currently underway under Phase II of the National River Health Program to determine the full implications of sampling and operator error for model sensitivity. This analysis includes quantifying error sources and their effects on the rates of misclassification to quality bands (*sensu* Clarke et al 1996).

Acknowledgments

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Chemical characteristics of stream waters in the Jabiluka region

C leGras, D Moliere & D Norton

Introduction

The physiography of the Jabiluka minesite embodies a significant paradox. This is that a world-class uranium (U) orebody is overlain by streams that contain the element at concentrations of only a few nanograms per litre, which is near the practical detection limit. Indeed, uranium concentrations ([U]) in Ngarradj Creek (Ngarradj) and its tributaries are in the bottom percentile of freshwater [U] worldwide (fig 1).

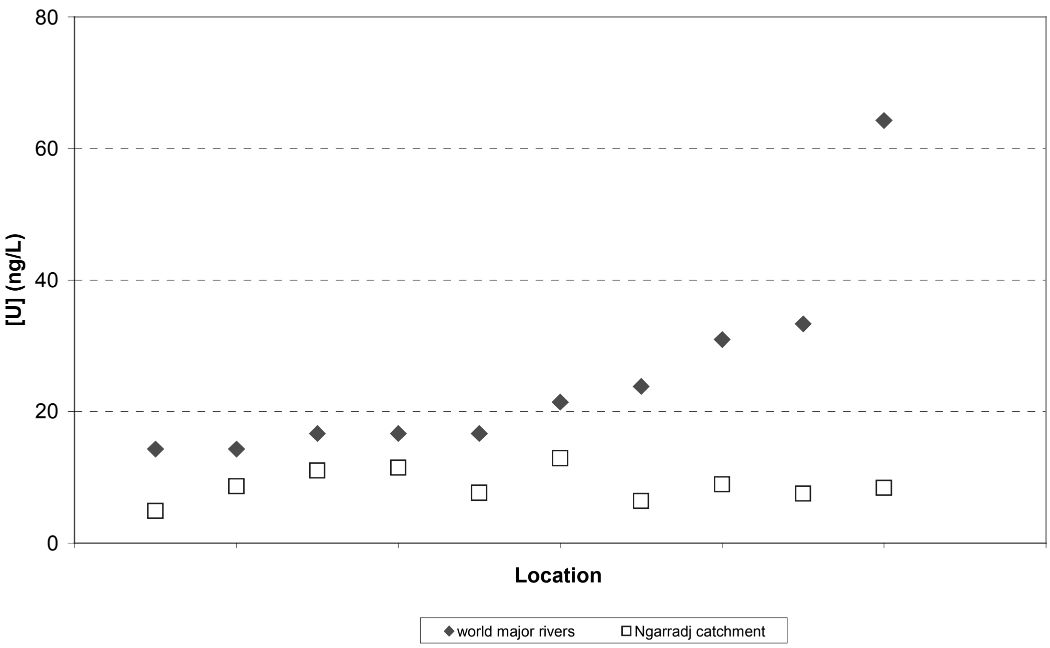

Figure 1 A comparison of mean [U] of various streams of the Ngarradj Creek catchment and the ten major world rivers with the lowest [U] (world data from Palmer & Edmond 1993)

The Jabiluka orebody also contains copper and lead values much greater than average crustal abundance (though below ore grade), but these metals also occur in stream water at concentrations that are frequently too low to measure.

The explanation for this situation lies in the local geology, where the orebody is hosted mostly by Lower Proterozoic graphitic schists, which are overlain unconformably by Middle Proterozoic sandstone and orthoquartzite (ERA 1996). The surface stratum is heavily leached, and even unweathered rock contains very low heavy metal contents (ERA 1996).

Therefore, the baseline and near-baseline data contained in this report provide a data set which will allow very small deviations from unaffected concentrations to be easily detected. This is an ideal position from a regulatory and monitoring viewpoint.

This report details critical indicator values in streams of the Ngarradj Creek catchment, and demonstrates the sensitivity of the sampling strategy that has been implemented. Two small streams in the immediate vicinity of the mine portal have been subjected to minor perturbations, probably with minor environmental consequences. Nevertheless, these perturbations are observed clearly and consistently, which increases confidence in the efficacy of the program. The main stream near the mine, Ngarradj Creek, shows no measurable effects from mining at present. This is despite the data being sufficiently sensitive and precise to demonstrate a high degree of intra-year and inter-year consistency for the indicators measured. This data set should therefore provide an adequate basis from which to observe small mine-related excursions from baseline, should they occur. For many indicators, notably U and sulphate (SO_4^{2-}), small spatial and temporal variations have been measured at concentrations that were below commercially accessible quantitation limits until recently.

A description of the sampling program and data set

The sampling approach

The sampling program commenced during the 1997–98 Wet season, and concluded at the end of the 2000–01 Wet season, yielding four years of data. Only data from the first three years are included in this report, except for qualitative reference to 2000–01 data where these are particularly relevant. A total of 31 sites have been sampled in Ngarradj Creek and its tributaries, and in a number of small creeks that flow westward from the escarpment outlier west of the minesite to the Oenpelli Road. These west-flowing stream sites constitute control samples. In addition, a further six sites were sampled in three adjacent catchments, also for comparison purposes.

A total of 21 physical and chemical indicators were measured at these sites, including general water parameters (pH, electrical conductivity-EC, alkalinity, organic carbon and turbidity), nutrients (total phosphorus and orthophosphate), major ions (chloride-Cl^-, SO_4^{2-}, magnesium-Mg and calcium-Ca) and heavy metals (aluminium-Al, cadmium-Cd, chromium-Cr, copper-Cu, iron-Fe, manganese-Mn, nickel-Ni, lead-Pb, U and zinc-Zn).

Most of these indicators are present in very low concentrations, at or near practical detection limits in some cases. These indicators are unable to provide meaningful spatial or temporal information, though the data are still useful as a basis for assessing mine-related deviations. Notable in this group is Pb, with a detection limit of 0.02 µg/L and with few measurements unequivocally above this. For this reason, Pb has not been discussed in this report. Many other indicators did not vary in a readily interpretable way, or else are not expected to be mine-related contaminants, and so have also been omitted from this report.

In the same way, the number of sites discussed has been restricted to those that would be most affected by mining activities, together with the corresponding control sites.

Mean values for important indicators

The broad overview of mean indicator values (table 1) shows that variation is relatively small throughout the whole suite. The main excursions are between the upstream and downstream sites of North and Central Tributaries. These small creeks define the northern and southern (respectively) boundaries of significant disturbance due to the mine. However, even though mine-related impacts can be inferred from these numbers, there exist substantial inter-year differences in the magnitude and temporal patterns of these indicator variations. These will be discussed in detail below.

Table 1 Mean indicator values at selected sites for the years 1997–98 to 1999–2000

Site	GPS location	pH	EC uS/cm	Turb. NTU	[Mg] mg/L	[SO_4^{2-}] mg/L	[Cu] µg/L	[Mn] µg/L	[U] µg/L
Ngarradj Creek upstream 1	132.931444 ºS 12.504000 ºE	4.74± 0.36	11.0± 2.1	1.1± 0.4	0.25± 0.07	0.32± 0.10	0.16± 0.09	3.2± 1.6	0.008± 0.003
Ngarradj Creek upstream 2	132.933940 ºS 12.503911 ºE	4.91± 0.22	11.4± 2.5	1.4± 2.1	0.24± 0.06	0.31± 0.20	0.18± 0.12	3.5± 1.6	0.008± 0.003
Ngarradj Creek downstream 1	132.921528 ºS 12.494194 ºE	5.37± 0.69	9.2± 4.3	6.2± 12.9	0.36± 0.09	0.23± 0.07	0.18± 0.06	3.5± 1.6	0.010± 0.002
Ngarradj Creek gauging station	132.922438 ºS 12.491447 ºE	5.33± 0.27	10.5± 3.0	2.1± 1.6	0.34± 0.12	0.24± 0.21	0.18± 0.11	3.4± 2.0	0.010± 0.004
Ngarradj Creek downstream 2	132.916667 ºS 12.484111 ºE	5.02± 0.58	13.0± 12.5	2.1± 0.8	0.39± 0.12	0.25± 0.13	0.18± 0.07	3.9± 2.3	0.011± 0.004
Ngarradj Creek (Oenpelli Road)	132.913628 ºS 12.467907 ºE	5.40± 0.26	11.2± 2.7	2.6± 2.2	0.36± 0.10	0.22± 0.14	0.20± 0.16	3.5± 1.3	0.011± 0.005
Ngarradj Creek west branch	132.927417 ºS 12.505722 ºE	5.83± 0.28	11.3± 7.8	5.9± 7.4	0.69± 0.49	0.16± 0.12	0.16± 0.07	3.2± 2.2	0.020± 0.011
East Tributary	132.932810 ºS 12.495093 ºE	4.97± 0.26	9.9± 3.1	1.3± 1.2	0.20± 0.06	0.22± 0.20	0.18± 0.15	2.6± 1.5	0.007± 0.002
Central Tributary causeway	132.915750 ºS 12.499653 ºE	6.05± 0.31	15.6± 4.0	2.0± 4.2	0.99± 0.27	0.08± 0.03	0.17± 0.17	3.5± 2.7	0.008± 0.004
Central Tributary downstream	132.911444 ºS 12.499361 ºE	6.10± 0.26	21.1± 6.3	3.0± 5.0	1.35± 0.50	0.11± 0.06	0.09± 0.09	6.2± 3.8	0.009± 0.007
North Tributary upstream	132.913712 ºS 12.498266 ºE	5.90± 0.28	8.5± 2.0	1.0± 0.6	0.47± 0.10	0.10± 0.05	0.16± 0.12	0.79± 0.40	0.007± 0.003
North Tributary downstream	132.915972 ºS 12.498556 ºE	6.12± 0.19	19.0± 15.6	4.3± 1.7	0.94± 0.78	0.27± 0.10	0.11± 0.12	2.8± 2.9	0.016± 0.011

Variation in indicator concentrations in Ngarradj Creek, North Tributary and Central Tributary

The physical and chemical character of Ngarradj Creek

Only small differences are evident between the six Ngarradj Creek sites sampled for the critical indicators Mg, Mn, U and SO_4^{2-}. The differences between the two sites upstream from the mine, and the four downstream sites are due mainly to the significantly different water chemistry of the West Branch of Ngarradj Creek. This is the largest tributary of the main channel and its confluence is between the two groups of sites. West Branch has higher [U] and [Mg], but lower [SO_4^{2-}] than Ngarradj Creek. Mn concentrations are almost identical, hence little difference in [Mn] is observed between the groups of sites. East, Central and North Tributaries, though with measurably different water chemistry from Ngarradj Creek, are either not sufficiently different or have too small a discharge to make an observable difference. Intra-year differences are more important than inter-year variations for SO_4^{2-} and

U (fig 2), with a pronounced 'washoff' effect consistently observed through the years. Except for the ordinate value, the figures are almost superposable. There is no evidence for any mine-related influence on water chemistry at any site in Ngarradj Creek.

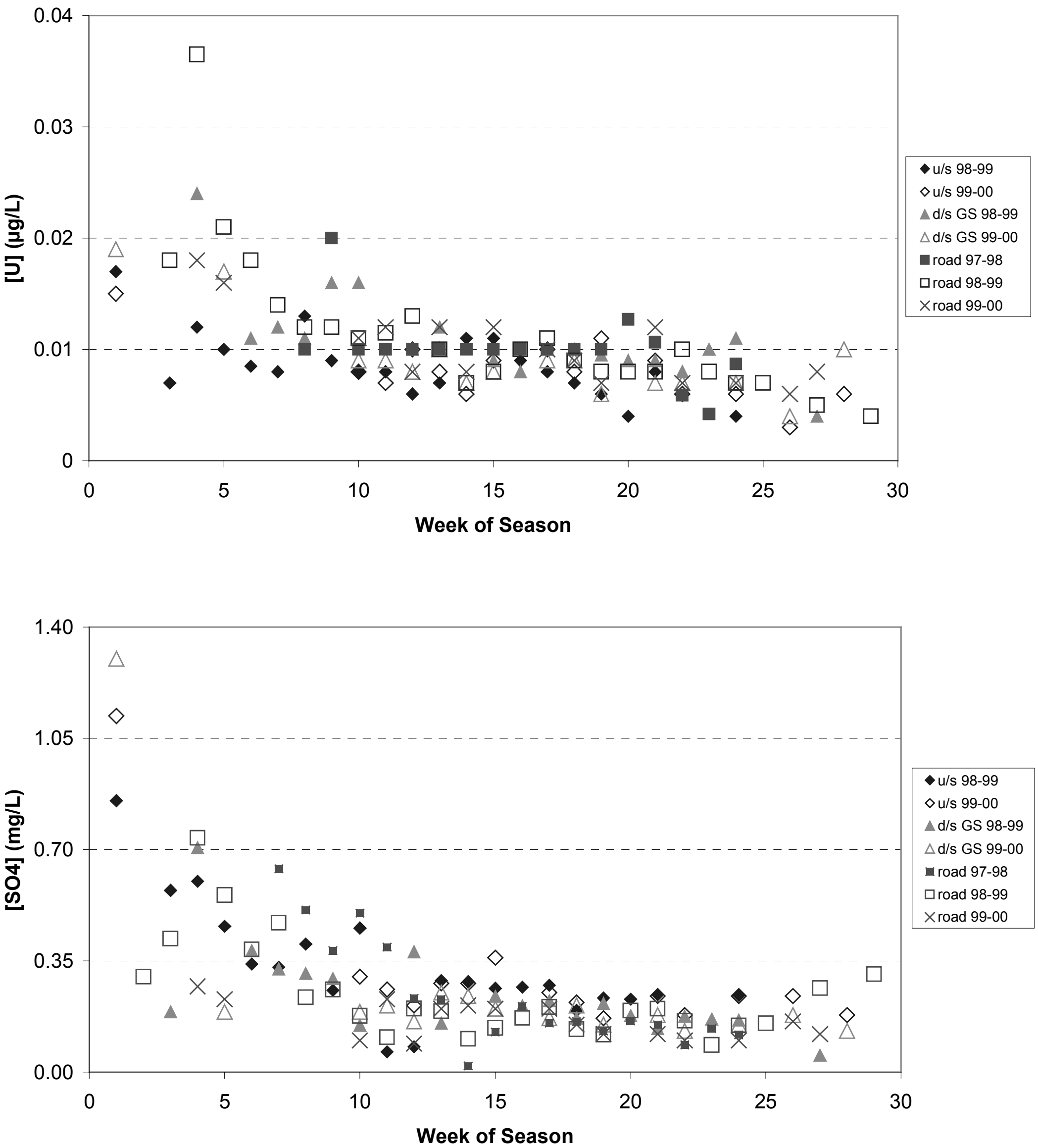

Figure 2 Spatial and temporal variation in [U] (top) and [SO_4^{2-}] (bottom) at three sites in Ngarradj Creek from 1998–2000

An interesting observation is the greatly divergent behaviour of U and Mn in Ngarradj Creek as a function of turbidity (fig 3). In this case, turbidity is used as a surrogate for discharge, for which detailed data are not available.

The best explanation for this dramatically different behaviour is that Mn is derived mainly from groundwater intrusion, and hence is present in higher concentration when hyporheic water forms a greater proportion of total discharge, that is, at low flow. Uranium, conversely is much more directly related to turbidity, which suggests that it is more closely associated with runoff. This interpretation accords with the expected redox behaviour of the elements, where Mn oxides should be readily reduced to Mn^{2+} in organic-rich shallow groundwater.

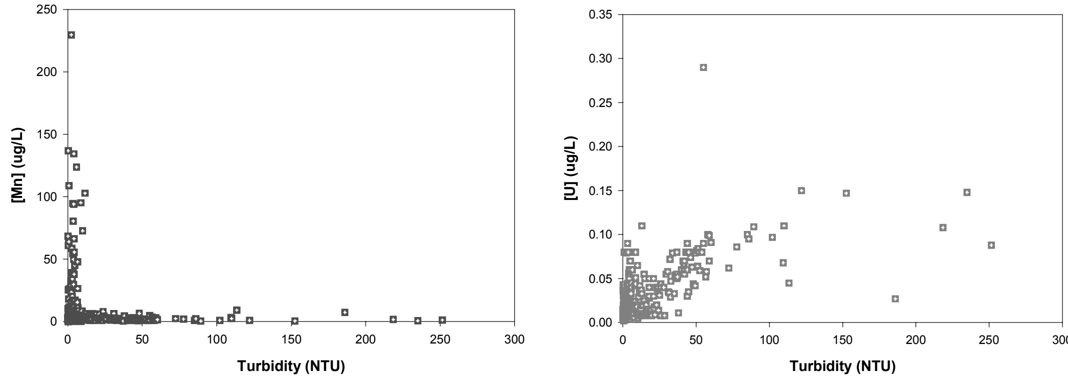

Figure 3 The relationship between [Mn] (left), [U] (right) and turbidity in Ngarradj Creek samples (all sites)

The physical and chemical character of Central Tributary

Two sites were sampled on Central Tributary. One site is immediately upstream from a vehicular causeway and is believed to be minimally affected by mine-related disturbance. The second site is about 500 m farther along, downstream from likely inputs from mine construction. Although there are no obvious sources of mine-related contaminants to Central Tributary, the water chemistry of the two sites is significantly different. This was not evident for the 1998–99 Wet season samples, the first after the commencement of mine workings. However, for the 1999–2000 samples, very evident differences were observed for [Mg], [Mn], [Ca] and pH, and to a lesser extent, [SO_4^{2-}]. Significantly, [U] does not change appreciably from year to year, averaging 0.008 µg/L at both sites during 1998–99, and being 0.009 and 0.010 µg/L respectively during 1999–2000. These trends have continued for the incomplete 2000–01 data set (not discussed further). The increase in pH suggests that the input source may be the dissolution of carbonate minerals (containing Ca, Mg and minor amounts of Mn). Increased SO_4^{2-} suggests that a small amount of sulphide mineralisation may have oxidised. The likely explanation is the partial weathering of waste rock used for construction purposes, although the mechanism of transport to Central Tributary is not immediately evident. Selected values are recorded in table 2. The generally higher values at the causeway during 1998–99 may be an artefact of the activity associated with construction.

Table 2 Selected indicator values for the causeway (upstream) and downstream sites on Central Tributary for the 1998–1999 and 1999–2000 Wet seasons

	[Ca] (mg/L)	[Mg] (mg/L)	[Mn] (µg/L)	[SO_4^{2-}] (mg/L)	pH
1998–1999	0.27 (causeway) 0.29 (d'stream)	1.1 (causeway) 1.1 (d'stream)	5.6 (causeway) 6.3 (d'stream)	0.08 (causeway) 0.08 (d'stream)	6.1 (causeway) 6.1 (d'stream)
1999–2000	0.04 (causeway) 0.47 (d'stream)	0.89 (causeway) 1.9 (d'stream)	1.8 (causeway) 6.0 (d'stream)	0.09 (causeway) 0.16 (d'stream)	5.7 (causeway) 6.0 (d'stream)

The physical and chemical character of North Tributary

In North Tributary, unlike Central Tributary, the reason for differences in indicator values between the upstream and downstream sites is evident. This is the large quantity of unmineralised orthoquartzite overburden that has been placed in the stream channel. The contents of target indicators in this rock are very low. This therefore allows a sensitive test of the ability of chemical testing to discern an impact on stream water quality from its presence.

A substantial difference was observed in the behaviour of common ions (Ca^{2+}, Mg^{2+} and SO_4^{2-}) and Mn as a group and U, as detailed in table 3.

Table 3 Selected indicator values for the upstream and downstream sites on North Tributary for the 1998–1999 and 1999–2000 Wet seasons

	[Ca] (mg/L)	[Mg] (mg/L)	[Mn] (µg/L)	[SO_4^{2-}] (mg/L)	[U] (µg/L
1998–1999	0.15 (upstream) 1.1 (d'stream)	0.48 (upstream) 1.3 (d'stream)	0.58 (upstream) 4.2 (d'stream)	0.11 (upstream) 0.31 (d'stream)	0.006 (upstream) 0.009 (d'stream)
1999–2000	0.04 (upstream) 0.25 (d'stream)	0.42 (upstream) 0.51 (d'stream)	0.74 (upstream) 1.0 (d'stream)	0.10 (upstream) 0.22 (d'stream)	0.006 (upstream) 0.024 (d'stream)

In 1998–1999 (the first Wet season after placement of the quartzite), electrolyte and Mn concentrations were much higher at the downstream site than in the succeeding year (1999–2000). This was particularly marked for Mn, where the concentration of this element progressively declined during the first year, as shown in figure 4, and did not return to previous, relatively high concentrations in 1999–2000. These observations suggest an initial washoff effect for this group of indicators.

Conversely, [U] is similar at both sites during 1998–99, but significantly higher at the downstream site in 1999–2000, as depicted in figure 5. This suggests that a period of initiation was necessary before measurable uranium values were released from the nominally unmineralised rock. The higher values are, however, only a factor of about five greater than the practical detection limit, and extremely low in world terms. These trends apparently continue in 2000–01, according to the incomplete data set for the current year.

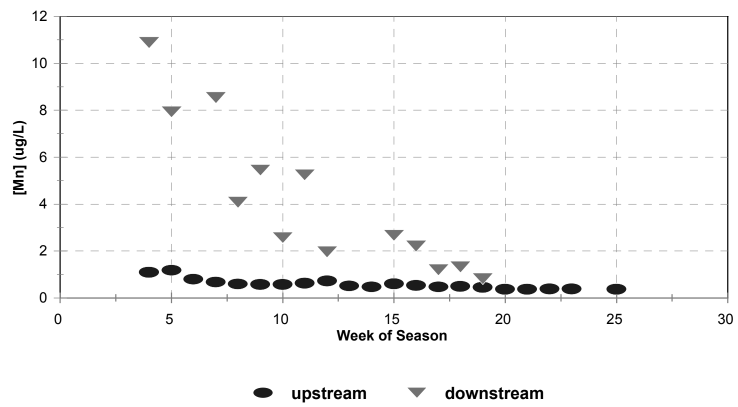

Figure 4 [Mn] in North Tributary during the 1998–1999 Wet season

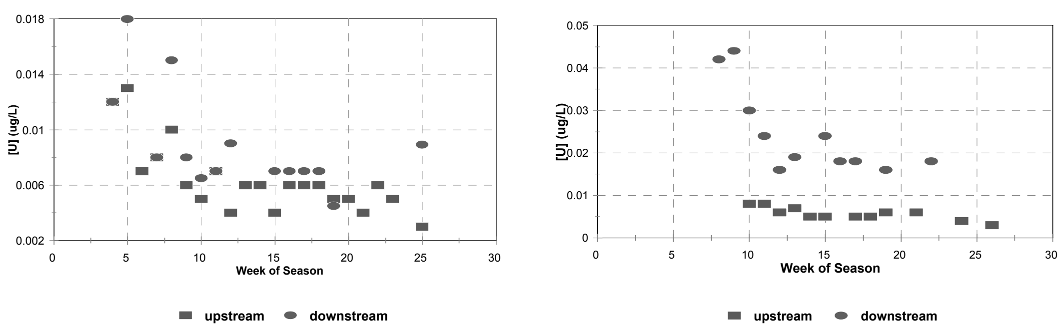

Figure 5 Comparison of [U] at the upstream and downstream sites of North Tributary in 1998–99 (left) and 1999–2000 (right)

Conclusions

The objective of the continuing project described here is to monitor a number of physical and chemical indicators in the streams draining the immediate vicinity of the Jabiluka mine. An original objective was to establish baseline indicator values at various sites. The main conclusion to date is that the concentrations of all measured indicators are very low, in some cases near or at the practical limits of detection. So far, the evidence from chemical monitoring suggest that the disturbance associated with mine construction has impacted minimally on proximate streams. Nevertheless, low-level impacts can be measured in creeks immediately downstream from sites of overt disturbance. These observations lend confidence to the expectation that physical and chemical monitoring will be able to detect significant changes in water quality that may be occasioned by further mine development.

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WETLAND ECOLOGY AND CONSERVATION

Ecology and Inventory

An analysis of wetland inventory and information sources[1]

CM Finlayson, NC Davidson[2,3], AG Spiers & NJ Stevenson[2,4]

1 Introduction

Over the past few decades many wetland scientists and managers have called for the collection of further information for wetland conservation and management. Knowledge of the location, distribution and character of wetlands, their values and uses, is required at a variety of geographical scales, ranging from local to national and international (Dugan 1990, Finlayson 1996a). Despite the compilation of such information many basic features of wetlands around the world have apparently not been recorded (Finlayson & van der Valk 1995).

Furthermore, the scattered nature of the information that is available has prevented effective assessment of the size and distribution of the global wetland resource (Finlayson & van der Valk 1995). Recognition of this situation by the Ramsar Convention on Wetlands led to the 'Global review of wetlands resources and priorities for wetland inventory' (GRoWI), a review of the state of wetland inventory and the extent to which it can yield information on the size, distribution and status of the global wetland resource (Finlayson & Spiers 1999, Finlayson et al 1999). The review was linked with analyses undertaken in two international workshops. One addressed practical issues for wetland inventory, assessment and monitoring (Finlayson et al 2001a); the other examined information management systems for assessing the conservation and status of wetlands (Davidson 1999). A summary of the review and these analyses is given below.

2 Definitions and concepts of wetland inventory

The global review of wetland inventory used an array of source material since an inventory was considered to be simply a collation of material on wetlands, specifically their location and size, possibly augmented with further information on their biophysical features and management. This broad interpretation of inventory was adopted after considering the report by Finlayson (1996a) who, after differentiating between a wetland inventory and a wetland directory noted that, in reality the terms were often used loosely and interchangeably.

However, due to inconsistency in usage it was necessary to differentiate between inventory, assessment and monitoring of wetlands. The definitions proposed by Finlayson et al (2001b) were adopted.

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1 More detailed discussion of this research is provided in Finlayson CM & Spiers AG (eds) 1999. *Global review of wetland resources and priorities for wetland inventory*. Supervising Scientist Report 144, Supervising Scientist, Canberra. & in Finlayson CM, Davidson NC, Spiers AG & Stevenson NJ 1999. Global wetland inventory — status and priorities. *Marine and Freshwater Research* 50, 717–727.

2 Wetlands International, Wageningen, The Netherlands.

3 Present address: Ramsar Convention Bureau, Gland, Switzerland.

4 Present address: Ecoscope, Cambridge, United Kingdom.

Wetland inventory — the collection and/or collation of core information for wetland management, including the provision of an information base for specific assessment and monitoring activities.

Wetland assessment — the identification of the status of, and threats to, wetlands as a basis for the collection of more specific information through monitoring activities.

Wetland monitoring — the collection of specific information for management purposes in response to hypotheses derived from assessment activities. (This definition is taken from that developed for the Ramsar Convention and Mediterranean wetland program; Finlayson 1996b,c.)

Thus, inventory, assessment and monitoring are separate processes, although inexorably linked. This position was also accepted by Costa et al (1996) when developing a protocol for wetland inventory in the Mediterranean. Finlayson et al (1999) and Costa et al (1996) also accepted that whilst inventory provided a basis for monitoring it was not itself a monitoring tool.

3 Information sources, collation and analysis

The information reviewed here is derived chiefly from the international projects noted above and complemented by information contained in draft protocols for an Australian national wetland inventory (Finlayson 1999), so providing a synthesis of the most up-to-date analyses available of inventory information for the world's wetlands. The most comprehensive of the international projects was the global review of wetland resources undertaken on behalf of the Ramsar Convention on Wetlands (Finlayson & Spiers 1999). This comprised separate reports for each of the Ramsar Convention's then seven geo-political regions (with a Middle East analysis reported separately from the remainder of the Asia region), a separate report on continental- and international-scale inventory, and a summary report covering results, conclusions and recommendations at the global scale.

The outcomes from the global review of wetland inventory were combined with information from the other projects to present an overview of the current status of wetland inventory around the globe. The overview was then used as a basis to outline and provide recommendations for standardised inventory methods, information management, and priority directions for improving further inventory.

4 Analysis of wetland inventory

Extent and coverage of wetland inventory

Of the 188 sources examined in detail only 18% were comprehensive wetland inventories; 74% were partial inventories, chiefly in two categories - either inventories of important wetlands only or inventories of particular wetland habitat types only; 9% were sources of information containing wetland inventory but covering a broader scope than solely wetlands, e.g. national soil surveys. Further, only 7% of the 206 countries and territories for which inventory information was sought have an adequate coverage of wetland inventory information (although several other countries do have major inventory programs underway or planned); 68% have only partial inventory coverage (important sites only and/or some habitat types only) and 25% of countries and territories have little or no inventory coverage.

The inadequacy of many inventories was also shown. Inventory methodology was not described in 18% of inventories. Only 30% of inventory sources defined the type of wetlands they included, and for over one-third (34%) no definition could even be inferred. Further, only 52% of sources provided a statement of the objectives of the inventory, and of these only half were undertaken specifically as baseline wetland inventory; and only 49% of sources provided area values per defined wetland type, although some others gave only overall wetland area (not divided by type). 35% proved to be compilations or reviews of existing material (undertaking no new data collection). There was variation in the extent to which inventories also provided information that is categorised as 'assessment' under the above definitions: half (49%) provided wetland status information; but only 21% contained wetland loss or degradation information; and only 8% consistent information on wetland values and benefits.

Extent and distribution of wetlands

Previous attempts to estimate the global extent and distribution of wetlands have been made through a variety of approaches. Such estimates vary considerably (5.6–9.7 million km^2 , Spiers 1999) and appear highly dependent on the type of source material and the definition of wetland used. The broad Ramsar Convention definition of a wetland has been used in many national wetland inventories (41%). There are also many inventories that have been restricted to more specific habitats, or exclude marine wetlands. However, in reviewing wetland inventory information this presented less of a problem than those that did not include a clear definition of the range of habitats being considered.

Spiers (1999) summarised global estimates for different wetland types (natural freshwater wetlands 5.7 million km^2; rice paddy 1.3 million km^2; mangroves 0.18 million km^2; and coral reefs 0.3–0.6 million km^2) and derived an estimate for these habitats only of 7.48–7.78 million km^2. As this estimate does not include a number of widespread and extensive wetland types, notably saltmarshes and coastal flats, seagrass meadows, karsts and caves, and reservoirs, it implies a considerably larger global area than previous estimates. Combining the estimates from the analyses of the individual Ramsar regions also gives a much larger minimum area than previous estimates, ~12.76 million km^2, for global wetlands (table 1). Since, as we have described above, these estimates are derived from extremely patchy inventory information, this implies that the global wetland area is considerably larger. However, given the patchiness of inventory these estimates must be treated as preliminary minimum estimates and indicative only.

Table 1 Regional minimum estimates of wetland area (from Finlayson et al 1999)

Region	Area (million km^2)
Africa	1.21
Asia	2.04
Eastern Europe	2.29
Western Europe	0.29
Neotropics	4.15
North America	2.42
Oceania	0.36

It is not possible to make an objective assessment of the various estimates given because many inventories merely repeat previously gathered information and/or do not clearly describe the accuracy and reliability of the data (Finlayson et al 1999). For example, few inventories have been regularly updated.

5 Standardisation of inventory approaches

The above analyses illustrate the inadequate standardisation of inventory techniques, including the means of recording and reporting the basic information that is necessary for determining, with confidence, the status of wetlands worldwide (Finlayson et al 1999). Many inventories lack basic information, notably the objective or purpose of the inventory, the wetland definition and classification system, the method(s) of data collection, source data for statistics of wetland area and wetland loss, name and affiliation of the compiler for individual site data, etc.

The development of a standardised and flexible framework for wetland inventory will help individual countries to prepare national wetland inventories in a format compatible not only with their objectives but also with the inventory of neighbouring countries. This would greatly improve the capacity for comprehensive wetland inventory on a national and ultimately global scale. Standardised approaches could be derived from existing models, notably those used in the Mediterranean (Costa et al 1996) and the United States (Wilen & Bates 1995). These approaches have been successfully adapted for use in other countries and could provide a basis for a standardised framework and wetland inventory database (Finlayson et al 2001b).

Finlayson et al (1999) also recommended the development of a hierarchical protocol to assist countries in undertaking their inventories cost-effectively through the use of a basic data set to describe the wetland. This should include the location and area and the basic features of the wetland that provide values and benefits to humans. This could include general indicators or descriptors of the water regime, water quality and biota. Adoption of standardised wetland classification systems would greatly assist in comparisons between sites and regions and provide a basis for management decisions that may lead to the collection of more specific information on threats, values and benefits, land tenure and management, and monitoring. Thus, it was further recommended that sufficient information (core data fields) should be derived to enable the major wetland habitats (at least) to be delineated and characterised at least at one point in time (table 2). Additional data fields could be added if required for specific purposes, such as wetland assessment, as recommended in draft protocols for an Australian wetland inventory.

6 Information management

The need to more effectively manage wetland inventory data has been outlined by Finlayson et al (1999). Even the maintenance of a minimum core data set requires considerable care and thought, at least in terms of accessibility and storage and software compatibilities. Finlayson et al (1999) stress that when inventory information is recorded it should be accompanied by clear records that describe when and how the information was collected and its accuracy and reliability. Such records were absent from many of the inventories reviewed in Finlayson & Spiers (1999).

As a minimum, a meta-database should be established and accessible. Regardless of the fields adopted it is essential that the meta-database follows an established data protocol and is readily accessible. The need to immediately develop a standard, versatile meta-database is increasingly widely recognised.

Table 2 Core and recommended additional data fields for wetland inventory and assessment (from Finlayson 1999)

Essential core data elements
Area and boundary A (size and variation, range and average values)
Location A (coordinates, map centroid, elevation)
Geomorphic setting A (where it occurs within the landscape, linkage with
Other aquatic habitats, biogeographical region)
General description (shape, cross section and plan view)
Soil (structure and colour)
Water regime (periodicity, extent of flooding and depth)
Water chemistry (salinity, pH, colour, transparency)
Biota (vegetation zones and structure, animal populations and distribution, and special features including characteristic or rare/endangered species)
Recommended additional information categories
Landuse (local and in the catchment)
Impacts and threats to the wetland (within the wetland and in the catchment)
Land tenure and administrative authority (for the wetland and critical parts of the catchment)
Conservation and management status of the wetland (including legal instruments and social or cultural factors)
Climate and groundwater features (noting that catchment boundaries may not correspond with those of groundwater basins)
Management and monitoring programs (in place and planned)

7 Priorities for future wetland inventory

Knowledge of the global wetland inventory resource is, on the whole, incomplete. All regions of the world - Africa, Asia, Oceania, Neotropics, North America, and Western and Eastern Europe - have information gaps and priority areas for wetland inventory. Some of these information gaps are urgent with the following being identified (Finlayson & Spiers 1999):

(1) Priority should be given to regions in which the wetlands are least known and considered the most threatened: areas where rapid population growth and development are combining with ineffective or non-existent wetland protection and sustainable-use legislation, to destroy and degrade wetlands at an alarming rate. The priority regions for further wetland inventory and wetland-loss studies are the Neotropics, Asia, Oceania, Africa and Eastern Europe.

(2) To make the task more manageable, priority should be given to encouraging countries that do not yet have a national wetland inventory to commit resources to complete one.

(3) Attention must also be given to the inventory of priority wetland habitats, targeting those for which there is little or no information, and those at greatest risk of degradation and destruction. Priority wetland habitats are as follows.

*Seagrasse*s. In southern Asia, the South Pacific, South America and some parts of Africa, seagrasses are under increasing threat.

Coralreefs. These are an important biodiversity resource that is under continuing threat.

*Salt marshes and coastal flat*s. These have generally not been included in wetland inventories, with few areal estimates and no true global picture being available. They are under increasing threat worldwide, particularly in Africa, Asia and Oceania.

Mangroves. Mangroves are better mapped than other coastal and marine wetlands, but serious inconsistencies exist and more comprehensive inventory is required. This should be used to better determine the mangrove loss that is proceeding at an alarming rate in many parts of Africa, south-east Asia and Oceania.

Arid-zone wetlands. These are poorly mapped but increasingly important in the light of escalating population pressures and water demand.

Peatlands. These are well mapped in comparison with other wetland habitats. However, they are threatened further in Europe, Asia and North America in particular, despite their importance as a global carbon sink and economic resource, and are poorly known in tropical regions.

Rivers and streams. Rivers and streams are seriously threatened in many regions of the world.

Artificial wetlands. These are increasingly important, with reservoirs, dams, salinas, paddy, and aquaculture ponds being important in many regions, notably Asia, Africa and the Neotropics.

(4) Steps should be taken to develop communciation between wetland users at all levels, from local to global, to ensure that the large amount of work required to establish, update or extend wetland inventory occurs. This is likely to require national action and a genuine will to identify key processes for targeted improvement.

(5) Co-operation between countries and agencies, with the common aim of improving wetland inventory for all wetland habitats, particularly those most threatened, should be enhanced.

(6) When undertaking further wetland inventory every effort should be made to link this with other national and international initiatives, such as the identification and delineation of further sites of international importance.

8 Recommendations

Eight priorities for action were recommended to the Ramsar Convention and accepted by its Contracting Parties in May 1999 (Ramsar Resolution VII.20).

(1) All countries lacking a national wetland inventory should undertake one, using an approach that is comparable with other wetland inventories and for which the Ramsar Convention should provide guidance.

(2) Quantitative studies of wetland loss and degradation are urgently required for much of Asia, Africa, South America, the Pacific Islands and Australia.

(3) Further inventory should focus on a basic data set describing the location and size of each wetland and its major biophysical features, including variations in area and the water regime.

(4) After acquisition of the basic data, further information should be collected with an emphasis on the management of threats to wetlands and uses of wetlands, land tenure and management regimes, and values and benefits of wetlands. Source(s) of information should be clearly recorded along with comments on its accuracy and availability.

(5) Each inventory should include a clear statement of its purpose and the range of information that has been collated or collected, the habitats covered and the date the information was obtained or updated.

(6) The Ramsar Convention should support the development and dissemination of models for improved globally applicable wetland inventory. These should be derived from existing

models and incorporate habitat classifications, and information collation and storage protocols.

(7) The Ramsar Convention should support development of a central repository for both hardcopy and electronic inventories. The meta-data that describe the inventories should be published on the World Wide Web for greater accessibility.

(8) Further support is required for completion of the global review of wetland resources and priorities for wetland inventory; and to develop procedures for regular updating and publishing of inventory information on the World Wide Web.

9 Further developments

The analyses reported here has led to the development of a 'Framework for wetlands inventory' for the Ramsar Convention, which will be considered for adoption at Ramsar's 8th meeting of the Conference of the Parties (COP8) in November 2002. The Framework provides a 13-step procedure for designing a wetland inventory, stressing that the precise nature of inventory chosen will depend on its purpose and objectives, geographical scale, and available resources. So as to assist in such choices, the Framework provides information on existing proven inventory methods, options for wetland classifications, a core recommended dataset, and an approach to identifying appropriate use of remote sensing datasets.

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Climate change and sea level rise in the Alligator Rivers Region, northern Australia[1]

CM Finlayson, I Eliot[2] & P Waterman[3]

Introduction

In 1995 the Intergovernmental Panel on Climate Change (IPCC) released its Second Assessment Report on the impacts, adaptations and mitigation of climate change with confirmed and dire projections about future scenario (IPCC 1995). (The Third Assessment Report was released in 2001 and confirmed such projections (IPCC 2001)). These reports noted the level of uncertainty with the scientific projections and the possible outcomes for both ecological and social systems. The assessments covered vulnerability to climate change of aquatic ecosystems and to water supply as a specific and important issue.

Coastal wetlands are expected to be highly vulnerable and susceptible to changes in the climate and sea level (IPCC 1995). Changes in temperature and rainfall, sea level rise, and storm surges could result in the erosion of shores and associated habitat, increased salinity of estuaries and freshwater aquifers, altered tidal ranges in rivers and bays, changes in sediment and nutrient transport and increased coastal flooding. Some coastal ecosystems are at particular risk, including saltwater marshes, mangroves and river deltas as well as non-tidal wetlands (IPCC 1995). The extent of this change will be influenced by the sensitivity, adaptability and vulnerability of the individual ecosystems and locations.

With this background we examined in 1996 the vulnerability of the large and valued freshwater ecosystems of the Alligator Rivers Region which includes Kakadu National Park, northern Australia, to determine if they were at risk from climate change and sea level rise (Bayliss et al 1998, Eliot et al 1999).

Climate change scenario

The major source of information for the prediction of climate change for this assessment was provided by Wasson (1992) and CSIRO (1994). Their projections are summarised in table 1. Scenarios for sea level rise were based on analyses adopted by the CSIRO (1994). Global projections of sea level rise range from 25–80 cm by the year 2100, with the most likely estimate being a rise of 50 cm. By the year 2030 sea level is expected to have risen an estimated 8–30 cm. The estimates require adjustment to allow for regional and site specific conditions to determine the relative sea level change at that place. Specific estimates are not currently available for Kakadu National Park or the wider Alligator Rivers Region.

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1 This paper is drawn from work published in SSR123: Bayliss B, Brennan K, Eliot I, Finlayson CM, Hall R, House T, Pidgeon R, Walden D & Waterman P 1997. *Vulnerability assessment of predicted climate change and sea level rise in the Alligator Rivers Region, Northern Territory Australia*. Supervising Scientist Report 123, Supervising Scientist, Canberra. Sections of it also appear in Eliot I, Waterman P & Finlayson CM 1999. Monitoring and assessment of coastal change in Australia's wet-dry tropics. *Wetlands Ecology and Management* 7, 63–81.

2 Department of Geography, The University of Western Australia, Perth, Western Australia, Australia (iane@sunny.gis.uwa.edu.au)

3 Environmental Management Services, Canberra, Australia (waterman@orac.net.au)

Table 1 CSIRO (1994) projections of climate change scenario for 2030 (adapted from Eliot et al 1999)

VARIABLE	PREDICTION
Temperature	Relative to 1990, northern coastal areas will be 1–2°C warmer.
Rainfall	Relative to 1990 rainfall in the NE summer rainfall region is expected to increase by 0–20% with a more intense monsoon.
Extreme events	Extreme events are expected to change in magnitude and frequency more rapidly than the averages – more hot days, floods and dry spells.
Clouds	Preliminary indication of an increase of 0–10% in total cloud cover in tropical areas.
Tropical cyclones	Cyclones could travel further south and their preferred paths may alter but effects on intensity are uncertain. ENSO could affect both the location and frequency.
Winds	Stronger monsoon westerlies are expected in northern Australia and stronger winds will accompany severe weather.
Evaporation	Anticipated 5–15% increase in potential evaporation by 2030.
Sea level	Best estimate for Australia for 2030 AD is about 20 $\pm$ 10 cm above 1990 levels with local variations in magnitude and frequency of extreme events, such as storm surges, waves and estuarine flooding.

Vulnerability assessment

The vulnerability assessment of the wetlands in the Alligator Rivers Region involved six steps in line with the procedure outlined by Kay and Waterman (1993). These were:

- Scoping the issues relating to climate and other changes in the region
- Identification of the natural, cultural, social and economic resources of the region
- Description of the biophysical change processes
- Assessment of the significance of predicted changes
- Determination of the range of responses to predicted changes
- Determination of the actions to be implemented by governmental and community agents

These analyses were aided by the existence of a substantial body of information that had been collected primarily for other environmental assessment purposes (see Bayliss et al 1998).

Resources potentially affected

Natural, cultural, social and economic resources across the Alligator Rivers Region could be affected by climatic and other changes. Specifically, sea level rise, shoreline erosion and saltwater intrusion could degrade both the salt and freshwater wetland resources. This would be manifest in:

- reduction or loss of some components of the mangrove fringe on the coast line;
- extensive loss of *Melaleuca* (paperbark trees) stands on the margins of some wetlands;
- colonisation of mangrove species along creek lines as an accompaniment to salt water intrusion; and
- replacement of freshwater wetlands with saline mudflats.

With changes in the wetland plant communities and habitats there would also be changes in animal populations, particularly noticeable would be changes to the community composition and distribution of bird species found in the freshwater wetlands. Additionally, there would be changes in morphology of the streams and billabongs and in the composition of the fish and other aquatic species. However, detailed analyses of habitat-species interactions have not been done. Changes in the natural vegetation and faunal resources may have cultural, social and economic consequences for the Aboriginal and non-Aboriginal people living in or visiting the area. The cultural resources have both social and economic resource values as they relate to the plants and animals used by the local Aboriginal people.

The cultural, social and economic resources that could be affected by accelerated change should be viewed as indicative of the breadth of factors to be considered, rather than exhaustive. Nonetheless, it serves to indicate the extent of possible changes to the resources of the park and the wider region.

Significance of potential change

There is a very substantial body of information describing geologic and especially, recent historic changes to the coast and wetlands of the region (see summary in Bayliss et al 1998). Oceanographic processes in the nearby marine ecosystems contribute to many of the changes and are manifested by high rates of shoreline erosion, changing tidal regimes within the river systems and contribution to saltwater intrusion into freshwater ecosystems. Changes resulting from these processes are seen in reduction of the fringing mangroves along the shoreline, expansion of the samphire and saltflat areas, colonisation of mangroves along estuarine levee banks, and the headward erosion of tidal creeks. However, the significance of such change has not been fully assessed.

Ecologic processes affected by environmental change include the expansion and contraction of plant communities with consequent effects on animal habitats. Again, insufficient knowledge of the interaction between wetland plant communities and changes in hydrological and depositional conditions makes prediction of the significance of long-term effects difficult. Wetland plant communities are viewed as being widespread in the region and highly dynamic in terms of variability in species composition, structure of the community and geographic spatial extent. The plant species are widespread at pan-regional and regional scales and no communities or individual species of rare or endangered species have been recorded. Similarly, animal species are widespread and no rare and endangered species are known from areas that could be affected by environmental change.

Pastoralists farming lands adjacent to the region have registered concerns over increasing encroachment of saline waters into freshwater wetlands that are used for seasonal pastures (Woodroffe & Mulrennan 1993). The potential loss of existing economic activities has been judged as significant given the attention directed towards remedial measures (Applegate 1999).

Management of a changing environment

Six broad environmental management issues were identified through the issue scoping process used for the vulnerability assessment. Many of the issues are common to the coastal margins of the Australian wet-dry tropics in general and underlie the possible management responses required to address the expected extent of ecological change in the wetlands. These issues and their implications are described below.

1 Perceptions and values

There has been no systematic examination of perceptions and values with respect to management of the region. However, societal perceptions and values manifest in the level of awareness of the possible effects of climatic and associated changes, as well as in attitudes held with regard to the hazards and threats to the environment resulting from climate change. Raising awareness of the implications of climate change is an important first step in changing governmental and community perceptions of the implications of climate change.

2 Hazard and risk

Natural hazards of the region include extreme weather events — tropical cyclones, monsoonal depressions, heavy rainfall, extended Wet seasons, excessively high temperatures and prolonged droughts; flooding, channel avulsion and bank erosion; inundation of coastal plains by storm surge; and coastal erosion, shoreline retreat, chenier migration and saltwater intrusion. Questions of responsibility and accountability may need to be addressed when changes due to particular hazards disrupt orderly use of coastal resources for habitation, industry and commerce, recreation and conservation.

3 Governance

Governance in the region and neighbouring catchments is currently not geared to deal with environmental change of the type and magnitude that is currently occurring. Issues are dealt with on a sectoral basis rather than in an integrated, intergovernmental and cross-sectoral manner (Finlayson et al 1998). However, environmental change is manifested across the biophysical region irrespective of jurisdictional boundaries. Governmental structures and community-based management mechanisms need to provide a consistent and appropriate response for system management, rather than simply addressing problems at a sectoral level.

4 Strategic management

Strategic management has two interrelated components, regional development and resource conservation. Regional development has strong economic connotations and raises questions about the best use of wetland areas. For example, the areas to the west of the Park are considered as important areas for seasonal pastures and measures have been proposed to prevent saltwater intrusion (Applegate 1999). In contrast, representatives from the fishing industry consider that the wetlands need to be retained as natural systems that support recruitment of commercial fish species. There is clearly a need for the broad community to resolve such conflict within the context of a regional development strategy that encompasses adequate conservation of resources.

5 Acquisition and custodianship of information

The acquisition and custodianship of information has been recognised as a key issue within Kakadu National Park as it is likely to impinge on:

- the strategic management of the responses to climate and other environmental change;
- research and monitoring needed to document the processes of change; and
- evaluation of the effectiveness of any management measures taken.

Lack of appropriate data and information causes poor decision-making and contributes to inappropriate management of coastal resources. An investment must be made in data and information with the object of reducing uncertainty, improving decision making, enhancing management capability and ensuring that unnecessary funds are not spent on ill conceived and poorly researched projects. Bayliss et al (1998) provide a summary of the information collated during the vulnerability assessment, in particular that from the wetland habitats most likely to be affected by climate change. This led to Eliot et al (2000) collating further data on the coastal environment and placing this within a management context. Whilst a large amount of data and information is available it has not always been possible to readily access this or use it for further assessment. Issues of data management have also been addressed and reported (Finlayson & Bayliss 1997, Eliot et al 2000).

6 Environmental research and monitoring

Ongoing environmental research and monitoring is required in the region to provide data and information for:

- further understanding of the processes and extent of environmental change;
- development of management strategies and action plans;
- implementing management prescriptions;
- auditing the effectiveness of management actions; and
- assessing performance of the overall management processes.

Research and monitoring should be broad in scope and include examination of social issues. The latter would include measures to raise the general level of awareness of natural variation in the environment of the region. Natural systems research is needed to document the processes of change and their effects on the biophysical environment. Such research will require a high level of innovation in order to integrate the cultural implications of change.

Requirements for management

As with broader environmental management a number of general principles can be applied when dealing with sea level rise. These include:

- avoid development in areas that are vulnerable to inundation;
- ensure that critical natural systems continue to function naturally; and
- protect human lives, essential properties and economic activities.

Though principles are applicable in the broader context of environmental change, their interpretation requires more detailed consideration of local factors and processes affecting environmental well being.

Institutional arrangements that are already in place for the region and adjacent wetlands (Finlayson et al 1998) may need to be developed to meet the governmental and community requirements for integrated management of the coastal wetlands. Current arrangements tend to focus on the specific issues confronting individual agencies or departments within a specific sphere of government. The intra- and inter-governmental dimensions of the issues identified for the coastal wetlands require an innovative approach because of the scale of the problems, and because they cross-jurisdictional boundaries.

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A review of wetland conservation issues in the Northern Territory[1]

MJ Storrs[2] & CM Finlayson

1 Introduction

The ecological character of wetlands of the Northern Territory (NT) has been described in a general sense and the major threats or management problems identified (Finlayson et al 1988, 1991, Finlayson & von Oertzen 1993). These reviews and reports on specific localities also identified major gaps in knowledge of basic ecological processes and threats to wetlands and, in the more isolated areas, even the character and extent of wetlands. Despite a general level of knowledge the information base is not uniform. Whilst reasonable data/information exists for some wetlands and/or threats to wetlands, a comprehensive inventory of all wetlands in the NT is not available.

A comprehensive review of wetland conservation and management issues for wetlands in the Northern Territory (Australia) was prepared by Storrs and Finlayson (1997) as a discussion paper for the development of a wetland conservation strategy. (A strategy was finalised and published by the Parks and Wildlife Commission of the Northern Territory (2000)). The conservation value of wetlands and the threats that they face were identified and described within a framework of sustainable utilisation of resources and maintenance of biological diversity. A summary of this paper is presented here.

2 Geographic setting

The NT lies between latitudes 11°S and 26°S an encompasses a large proportion of tropical Australia. The 1 347 x 10^6 km^2 area of the NT is divided into three broad landforms: i) the tropical northern zone that contains large rivers with wide coastal plains with permanent and seasonal wetlands; ii) the central semi-arid zone of uncoordinated drainage that contains seasonal and intermittent wetlands; and iii) the southern lowland zone of coordinated drainage that contains intermittent and episodic wetlands.

The NT comprises two broad climatic zones. In the north, the warm-hot wet season commences late in the year (Nov-Dec) and lasts for 3–4 months. The remainder of the year is cooler and mostly dry. South-easterly trade winds and a high pressure belt dominate the climate in the south. During the warm months (the temperate summer months) the south-east trade winds are interrupted by intrusions of moist air from tropical low-pressure troughs to the north. Most of the annual rainfall comes from violent convectional thunderstorms. Overnight frost can occur at sites in the south.

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1 A more detailed discussion of this topic is found in Storrs MJ & Finlayson CM 1997. *Overview of the conservation status of wetlands of the Northern Territory*. Supervising Scientist Report 116, Supervising Scientist, Canberra.

2 Northern Land Council, PO Box 42921, Casuarina, NT 0811, Australia.

3 Current knowledge of wetland resources

For convenience of summarising the information on wetlands of the NT the geographic area has been divided into three broad administrative regions — northern, central and southern regions. Given difficulties with systems used to classify wetlands a simplistic and interim classification was adopted for this review (table 1). Storrs and Finlayson (1997) also provide maps that show the distribution of plants species generally associated with wetland habitats.

Table 1 A simplistic and interim classification system for wetlands of the Northern Territory. The general occurrence of wetland categories within the three broadly delineated regions of the NT is also shown.

Wetland categories		Region of the NT
1	Coastal salt marshes	Northern/Central region
2	Mangrove swamps	Northern/Central region
3	Freshwater lakes and swamps	Northern/Central/Southern region
4	Floodplains	Northern region
5	Freshwater ponds	Northern/Central/Southern region
6	Seasonal and intermittent saline lakes	Southern region

It was not possible, based on current inventory information, to accurately depict the extent of wetlands across the NT. Maps produced by Paijmans et al (1985) whilst not specifically delineating discrete wetlands were used to illustrate a number of key points about the distribution of wetlands in the NT that reflect the general landforms and climate:

- general low occurrence of permanent swamps and lakes;
- permanent and near permanent wetlands occur along the coast and in the northern area;
- episodic lakes and land subject to inundation are spread across the central and southern regions; and
- generally dry wetlands occur across most of the central and southern regions

Storrs and Finlayson (1997) list the many major datasets on wetland features held by NT government agencies and used this along with past reviews of northern Australian wetlands to describe the major known features of wetlands in the NT.

4 Conservation status of wetlands

Generic comments on the conservation status of wetlands in the NT and the pressures that they face are described below. It is stressed that further assessment, monitoring and even audit of the conservation status of the wetland habitats and ecosystems is still needed.

Invasive plants

Major weed species include *Acacia nilotica* (prickly acacia), *Cenchrus ciliaris* (buffel grass), *Eichhornia crassipes* (water hyacinth), *Salvinia molesta* (salvinia), *Parkinsonia aculeata* (parkinsonia), *Prosopsis limensis* (mesquite), *Tamarix aphylla* (athel pine), *Brachiaria mutica* (paragrass), *Echinochloa polystachya* (aleman grass), *Hymenachne amplexicaulis* (olive hymenachne) and *Mimosa pigra* (mimosa).

Paragrass and other pasture species such olive hymenachne, present a particularly difficult problem given that pastoralists desire them while conservation and fisheries authorities are concerned over their potential to alter the ecological character of wetlands. Paragrass is highly invasive and has spread across many wetlands in northern Australia, aided by deliberate plantings. The major weed species is undoubtedly mimosa which has spread across coastal floodplains in an arc extending from the Moyle River in the west to the Arafura Swamps in Arnhem Land (Harley 1992). It covers an estimated 80 000 ha. Research efforts have centred on finding suitable biological control agents with a number having been released. Integrated control programs are also in place and incorporate biological control along with the use of herbicides, mechanical removal (chaining), burning and revegetation.

Invasive animals

Major invasive animals include *Bubalus bubalis* (Asian water buffalo), *Sus scrofa* (pig), *Bufo marinus* (cane toad), *Equus caballus* (horse) and *Equus asinus* (donkey), *Camelus dromedarius* (camels), and *Oryctolagus cuniculus* (rabbit). Prior to the 1980s feral Asian water buffalo proliferated on the coastal floodplains of the NT and were considered responsible for widescale destruction of the native vegetation by direct grazing, trampling and wallowing, and indirectly by destroying levee banks and contributing to premature drainage of freshwaters (Finlayson et al 1988). However, throughout the 1980s the feral herds to the west of Arnhem Land were almost eradicated as part of a national program to prevent diseases being transferred to domestic stock. They still exist in large numbers in Arnhem Land.

Feral pigs are widespread and have caused widespread damage around the edges of wetlands. There is also evidence they have proliferated following the removal of the buffaloes from the floodplains (Corbett 1995). Control of pigs is widely regarded as difficult in certain types of terrain. Camels concentrate around salt lakes and clay pans in the southern region of the NT while horses and donkeys are prevalent in the southern and central regions. The extent of their impact on wetlands is unknown. The impact of grazing by rabbits severe in the southern region and tapers off northwards. Excessive grazing can devastate the vegetative margins of ephemeral lakes and pools. Of increasing concern are the cane toads that have moved westwards from Queensland into the NT. There is major concern and uncertainty about the effect they will have on native fauna.

Fire and burning regime

Fire is a conspicuous element of the landscape in the northern part of the NT and burning patterns have changed considerably in recent decades. Andersen (1996) questions the emphases of some fire management regimes and along with other authors points out that the ecological consequences of burning patterns are, on the whole, inadequately known. Roberts (1996) refers to the wealth of traditional Aboriginal knowledge on fire and burning regimes in relation to food availability. In some landscapes there is a deliberate policy that attempts, amongst other objectives, to re-establish some semblance of traditional Aboriginal burning (Ryan et al 1995, Roberts 1996). Vast areas of the central and southern regions of the NT are also burned on a regular basis, including many intermittently or episodically flooded wetlands.

Overgrazing

Soil erosion, due largely to poor land management including overgrazing has resulted in extensive degradation of waterholes, stream banks and the riparian vegetation (Winter 1990). Heavily grazed wetland communities tend to converge floristically and introduced pasture species are known to replace the native grasses (Liddle & Sterling 1992). In addition to changes in the vegetation changes in primary production also have an adverse effect on fisheries production in estuaries (Griffin 1996). Overall, however, little is known about changes due to grazing.

Tourism and recreational activities

The environmental impact of tourism and recreational activities on wetlands has not on the whole been specifically investigated. The notable exceptions are possibly the effect of boats in specific locations, the pressure of recreational fishing on barramundi (*Lates calcarifer*) stocks, and the hunting of geese by non-Aborigines. However, the pressure from such activities needs to also consider infrastructure and associated disturbance. Anticipated increased tourism and extension of recreational activities is expected to increase pressure on wetlands. Hunting of geese by non-Aborigines has recently undergone increased regulation and has been subjected to intensive research and monitoring.

Pollution and contaminants

Water pollution, in particular that associated with mining has received a large amount of attention. The former mining activities at Rum Jungle are the subject of extensive rehabilitation effort while the Ranger uranium mine is the subject of a strict testing and monitoring regime (Humphrey et al 2000). Pesticides and fertilisers are used extensively in agricultural projects and for weed control on coastal wetlands, but little information is available on their impact on wetlands. Pollution from sunscreens, soaps and insect repellents may become a problem in small permanent waterholes frequented by tourists, as could fuel spillage from boats. Lead poisoning of waterfowl from ingested shotgun pellets is a problem at hunting reserves in the northern region (Whitehead & Tschirner 1991). Salinity is possibly the major concern in the coastal freshwater wetlands (Jonauskas 1996). Bayliss et al (1997) consider that many of these wetlands are under threat from sea level rise in association with the Greenhouse Effect.

Water regime and physical modification

Water regulation and physical modification of wetlands in the NT occur, but not to the same extent as in eastern Australia. Small barrages and dams are being constructed, but at this stage these are not considered to be excessively detrimental, although at times these issues have been hotly debated (Julius 1996). The clearing of mangroves for port, industrial and/or residential purposes has also aroused controversy. The proposed development of irrigated agriculture has raised concerns over water logging, sedimentation and discharges of polluted water.

5 Current uses of wetlands

The NT Government has adopted a policy of multiple land use for wetland management and encourages different land uses in balance with conservation objectives. Current uses include: pastoralism, grazing and some horticulture, commercial fishing, tourism and recreation, especially amateur fishing, crocodile egg harvesting, commercial pig harvesting, safari style buffalo hunting, conservation and nature reservation, and traditional subsistence. Land uses are more intensive in the seasonally inundated and very productive wetlands near Darwin in the northern coastal region than in the wetlands of the semi-arid and arid areas.

Pastoralism has been by far the most extensive land use in the NT. The wetland areas are the most nutrient rich and mesic areas and thereby produce the best forage for livestock.

However, there has been much debate on the efficacy of pastoral activities in the arid zone of Australia with periodic calls for the removal of grazing from at least some areas. Tourism and recreation are increasingly important land uses based on natural and cultural values. The recreational fishing industry is well established in the coastal wetlands with barramundi being favoured species (Julius 1996, Griffin 1996). Commercial fishing also occurs with barramundi and mud crabs being targeted.

New commercial uses of wetlands are being developed, such as wildlife utilisation (eg goose and crocodile egg collection), expanded tourism (eg hunting or photographic safaris or cultural and wildlife tours) agriculture and horticulture. Many of these activities are being investigated on wetlands owned by Aboriginal people.

Major issues

Based on information collated in the review a number of generic strategic issues for successful management of wetlands in the NT have been identified (Table 2). These have been expressed within the context of the specific objectives of the Conservation Stategy for the NT (NT Government 1994).

Table 2 Strategic issues for wetland conservation in the NT

Conservation strategy objective	Strategic issues for wetland conservation
Understanding	Develop and maintain a comprehensive inventory database for all wetlands, including protocols to ensure it is updated at regular intervals
	Characterise and quantify the importance of the physical and ecological interactions and linkages that occur between wetlands
	Characterise the processes that maintain the ecological character and values and quantify the importance of ecological benefits of wetlands
Public awareness	Develop community awareness of the extent, values and benefits of wetlands.
Protection and management	Implement catchment-wide land use planning processes that encompass wetlands and ensure the maintenance of their ecological character and values and benefits
	Enhance the reservation and management of wetlands within a systematically developed protected areas network that is representative of the diversity of wetland habitats, species and values and benefits
	Instigate specific management arrangements for wetland conservation and sustainable utilisation regardless of land tenure
	Enhance the level of control over and planning of grazing activities, especially in the central and southern regions
Monitoring	Develop and implement monitoring programs that describes the ecological character of wetlands to provide early warning of any potential adverse change.
Restoring	Assess the extent of ecological degradation caused by specific pest species and develop appropriate control measures based on rigorous risk assessment.
	Undertake immediate control measures for effective management of salinisation and other effects associated with climate change, especially in the northern region.
	Undertake immediate control measures for effective management of grazing in all wetlands.
Reviewing	Develop and implement a regular and systematic reporting process on the state of wetlands.

In developing these strategies a large emphasis was placed on integrated land use and planning policies. It is doubtful that these could be effectively developed and implemented without the adoption of an integrated and multi-sectoral approach to wetland management.

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Ecological character of two lagoons in the lower Volta, Ghana[1]

CM Finlayson, C Gordon[2], MJ Storrs[3] & Y Ntiamoa-Baidu[4,5]

Introduction

The socio-economic benefits of coastal lagoons in Ghana to the local people are very apparent — large quantities of fish and crabs are caught and traded; reeds and other plants are cut for thatch and for weaving mats; vegetables are grown in sandy garden beds irrigated by water drawn by hand from wells along the edges of the lagoons; and salt is extracted by both intensive and extensive methods. In recent years, the immense conservation value of the lagoons has also been recognised (Ntiamoa-Baidu & Grieve 1987, Ntiamoa-Baidu 1991, 1993, Piersma & Ntiamoa-Baidu 1995, Ntiamoa-Baidu et al 1998).

However, it became evident to authorites in Ghana, that the values and benefits provided by the lagoons were under increasing threat from over-exploitation and degradation. In response to this situation the Ghana Coastal Wetlands Management Project (CWMP) was implemented by the Ghana Wildlife Department (now Wildlife Division of the Forestry Commission) as part of the Ghana Environmental Resource Management Project, funded by the Global Environmental Facility (GEF). The general aim of the CWMP was to create an enabling environment so as to manage five coastal wetland sites. The basic rationale for the project was to maintain the ecological integrity of the lagoons and to enhance the benefits derived from the wetlands by local communities.

In support of this aim we provide a description of the ecological character of the Keta and Songor lagoons in the Lower Volta region of Ghana (fig 1). The detailed information collected in this study is presented by Finlayson et al (2000). Here we provide information on the climate, hydrology, chemistry, and the aquatic/wetland invertebrate fauna and the flora, and identify the major threats to the sustainable use of the lagoons.

Methods

Sampling of each lagoon and the surrounding wetland vegetation was based on a stratified grid drawn at intervals of 1' latitude and longitude (ie ≈ 1.8 x 1.8 km). The points of intersection of the grid were used as the basis for selecting sites for sampling. Samples were also collected from the Angor channel that connects the Keta lagoon with the Volta estuary.

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1 More detailed discussion of this research is provided in Finlayson CM, Gordon C, Ntiamoa-Baidu Y, Tumbulto J & Storrs M 2000. *The hydrobiology of Keta and Songor lagoons: Implications for coastal wetland management in Ghana*. Supervising Scientist Report 152, Supervising Scientist, Darwin.

2 Volta Basin Research Project & Zoology Department, University of Ghana, Legon, Accra, Ghana.

3 Formerly ***eriss*** (current address: Northern Land Council, PO Box 42921, Casuarina, NT 0811, Australia).

4 Zoology Department, University of Ghana, Legon, Accra, Ghana; Ghana Wildlife Society, Accra, Ghana.

5 present address: WWF International, Avenue du Mont-Blanc, 1196 Gland, Switzerland.

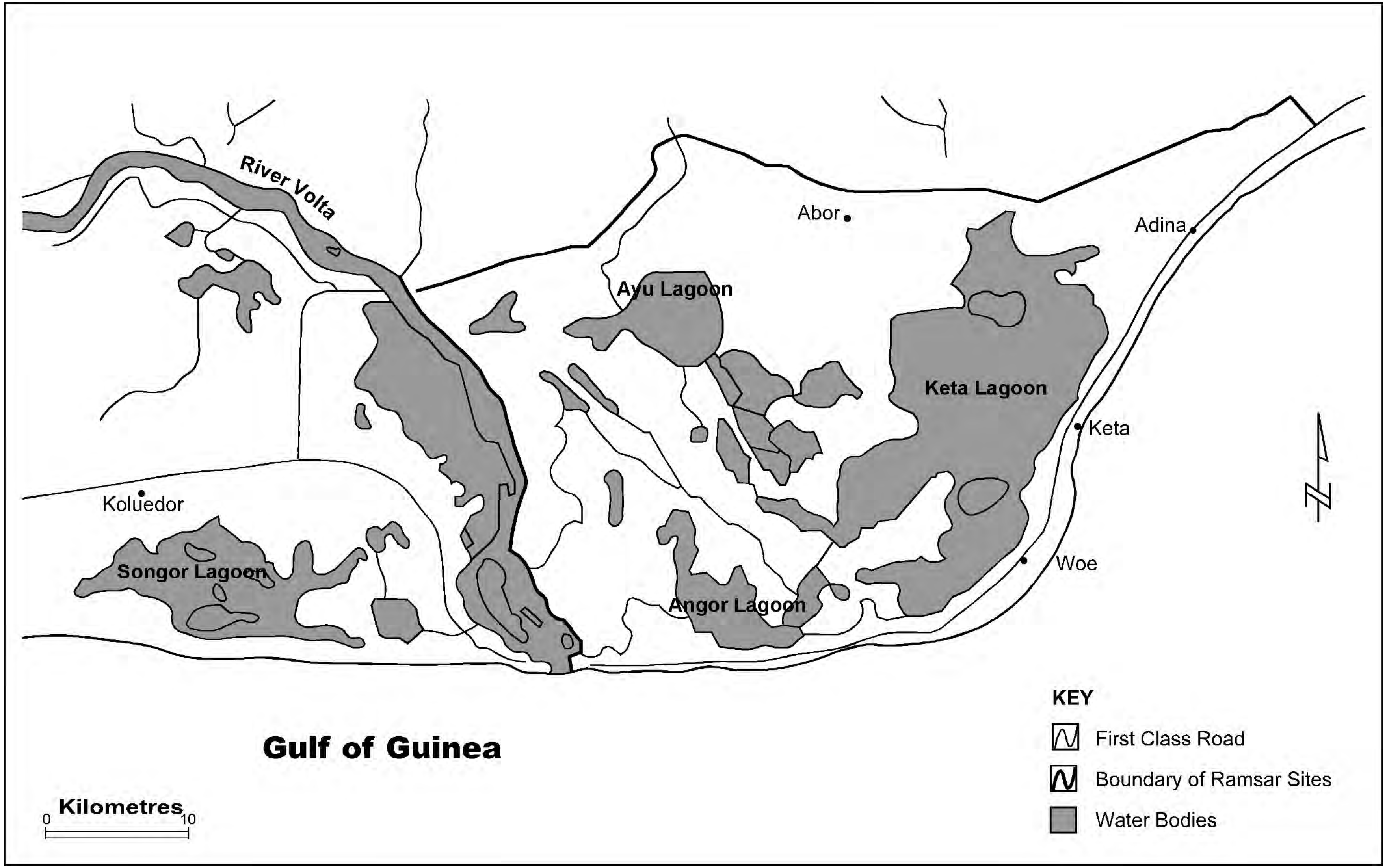

Figure 1 Location of Keta and Songor lagoons alongside the lower Volta River in eastern Ghana

For logistic convenience, the sampling strategy was divided into two components — one aquatic and the other wetland/terrestrial. All intersecting grid points within the lagoons were used for aquatic sampling (ie physico-chemical and biological parameters). The wetland sampling was based on a series of grid points located along the landward side of the lagoon shorelines. The details of the sampling strategy and location of all sites are shown in Finlayson et al (2000).

Ecological character of Keta and Songor lagoons

Climatic conditions

The study area lies within the dry equatorial climatic region of coastal Ghana which has two clearly defined seasons, the Dry season and the Rainy season. The Rainy season exhibits two peaks of rainfall, the main one occurring between April and June and the minor one between September and October. June is normally the wettest month. The mean annual rainfall is 892 mm, although it has been lower in the past decade. The prevailing wind direction is from the southwest (the southwest monsoons). Mean monthly averages of daily wind speed range between 21.1 to 29.0 km h^{-1}. However, high velocity winds (110 km h^{-1}) of short duration have been recorded in Accra. Average temperatures range between 23ºC and 31ºC with August usually being the coldest month.

Hydrological conditions

In general, stream flow in the area is seasonal, and corresponds to the seasonal variation in rainfall. A few coastal streams drain the area above the Keta lagoon which is about 24 km long and 12 km wide with an area of 272 km^2 and volume of 5 560 270 m^3 with water depths ranging from 0.08–0.75 m. The major streams apart from the Volta River include the Todzie River, which discharges into the Avu lagoon just north-west of Keta lagoon, and the Belikpa River, which discharges directly into the Keta lagoon. The Keta lagoon is connected to the Volta system by the Angor channel which is currently being dredged due to blockage by weeds and sediments.

The Volta River drains two-thirds of the country and is dammed first at Akosombo and then at Kpong. Before 1964, the year the dam at Akosombo was created, records on the Volta at Sogakope showed that water levels increased from 1.4 m in the Dry season to about 6.6 m in September or October. Discharge also varied between over 10 000 m^3 s^{-1} after the rains to less than 50 m^3 s^{-1} in the Dry season. After the construction of the Akosombo dam, however, water levels and flows were more uniform.

A few coastal streams drain the area above the Songor lagoon. The Sege River has a catchment area of about 75 km^2 and drains the north-western part of the Songor lagoon. There are no flow records for this river. The other major stream draining into Songor lagoon flows through Hwakpo. Songor lagoon covers an area of 115 km^2 and extends for about 20 km along the coast and 8 km inland behind the narrow sand dune. In general the lagoon normally dries in the Dry season and a sand dune is physically broken to allow seawater to flow into the lagoon at high tide. It is subsequently closed and the water evaporates under natural conditions throughout the year. The part of the lagoon, which is not managed for salt extraction mixes with freshwater from the catchment and undergoes natural evaporation until it dries completely.

Water chemistry

The pH of the water in Keta lagoon was neutral to alkaline (6.7 to 9.7 range) and in Songor slightly acidic to alkaline (5.2 to 8.3). The fairly high carbonate content of the water (generally 100 to 170 CO_4 mg L^{-1}) may have buffered any pH changes that could have resulted from biotic activity. Temperature in these shallow lagoons was always high, the mean often exceeding 30ºC. There was very little temperature difference between the surface and bottom. Unlike other water bodies in Ghana the water was not super-saturated with dissolved oxygen. The surface waters are generally well oxygenated throughout the night with lower levels at depth. It is assumed that wind action plays a greater role than photosynthesis in establishing the dissolved oxygen profiles.

The water was also basically without true colour. However, due to the strong wind action and the shallow nature of the lagoons, the transparency was often reduced to less than 10 cm. The transparency was reduced in areas where there was a large clay fraction in the sediment and high in areas where there were submerged aquatic plants.

Conductivity values range from under 2 mS cm^{-1} to over 80 mS cm^{-1}. The presence of hypersaline subsurface water was noted in several areas, especially in Songor lagoon. Sodium and chloride dominate the ionic composition of the water in Keta lagoon ranging from 4373 ± 1788 mg L^{-1} and 10 207 ± 8527 mg L^{-1} respectively. This is to be expected given the proximity to the sea. The trace metal concentrations in the water in Keta lagoon was usually below the limits of detection (0.03 mg L^{-1}) for lead and copper and ranged from 0.04–0.09 mg L^{-1} for zinc. Phosphorus concentrations ranged from 0.03 to 0.10 P-PO_4 mg L^{-1}.

Aquatic ecology

The phytoplankton consisted primarily of benthic diatoms that have been dislodged from the bottom of the lagoons, plus a few true planktonic diatom species. These species are typical of shallow lagoons and most likely make a significant contribution to the primary production. Some blue-green alga species made up the remainder of the biomass. Both lagoons harboured diatoms characteristic of high salinity close to seawater or even higher. The only freshwater or brackish water species present were found in the samples from the Angor channel connecting Keta lagoon to the Volta. The two lagoons are characterised by separate diatom assemblages, although many species are common to both.

The total chlorophyll concentrations in Keta ranged from undetectable to 145 μg L^{-1} with a mean of 20 ± 21 μg L^{-1} and in Songor lagoon from undetectable to 86 μg L^{-1} with a mean of 24 ± 19 μg L^{-1}. In Keta chlorophyll *a* was generally present in greater concentrations than *b* or *c* whereas in Songor chlorophyll *b* was more prevalent in some, but not all sites. These figures suggest that the lagoons are highly productive.

As would be expected from temporary waters with extremes of salinity and temperature, the zooplanktonic diversity in the two lagoons was not high. The situation was further complicated by the shallow nature of the waters sampled, leading to the appearance of several epibenthic species in the water column. Three main groups were found in the plankton sampling: Ostracods, Copepods and Amphipods. Zooplankton were found in all parts of the lagoon.

The macroinvertebrate fauna was dominated by three groups of organisms: annelids, molluscs and crustacea. For this type of waterbody, the insecta seemed under represented. This may be due to the large numbers of fish that are found in the lagoons and the salinity of the water. In Keta lagoon the mollusc *Tivela* sp. was most common in the macro-benthos followed by the gastropod *Tympanotonos* sp. In Songor, polychaetes were the most common organisms. The macro-zoobenthos reached remarkable numbers at some sites, oligochaetes were found in numbers exceeding 70 000 per m^2. The distribution and relevant abundance of *Nereis* and *Tympanotonos* are presented in figures 2 and 3.

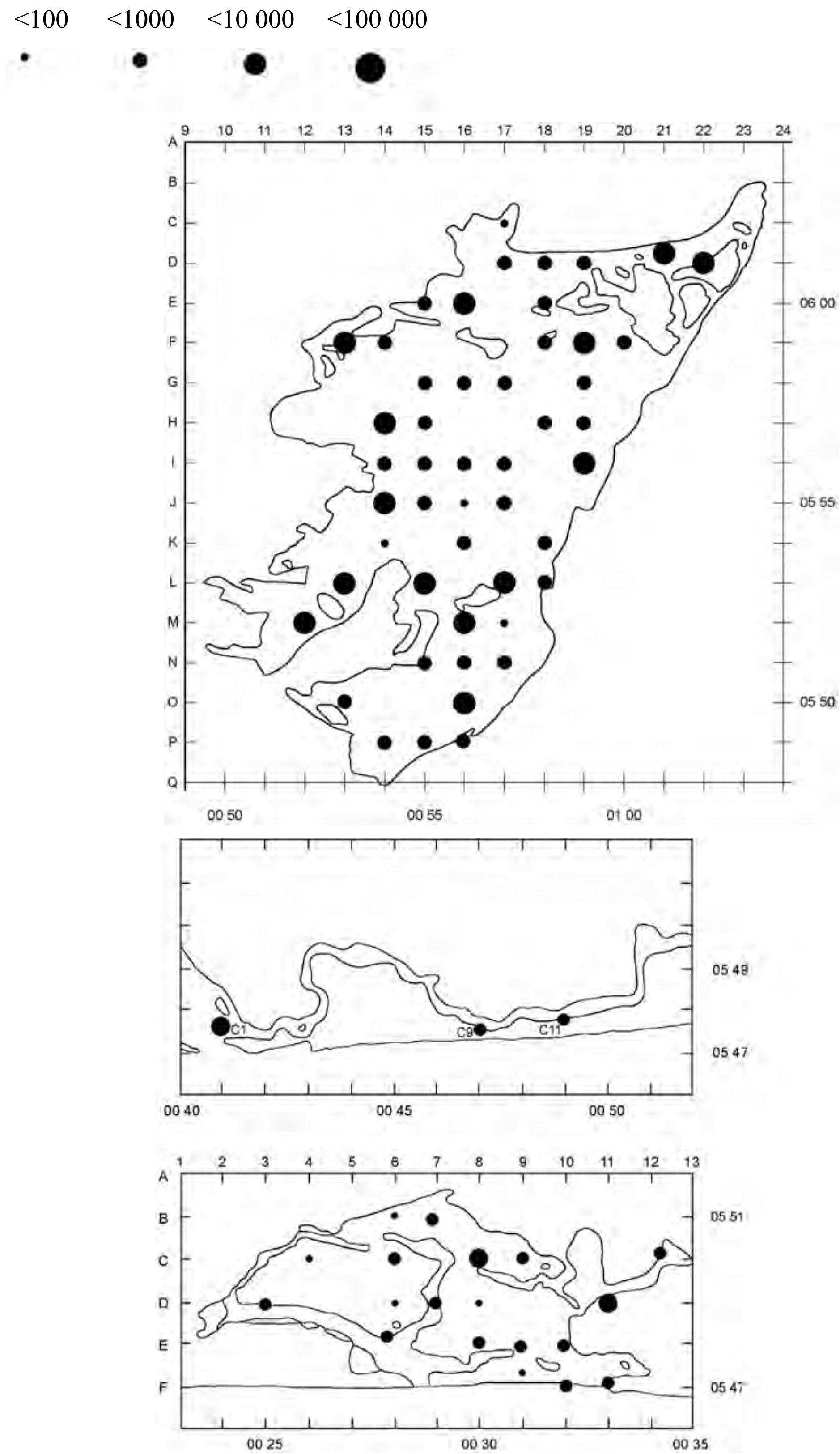

Figure 2 Distribution and abundance of *Nereis* sp. in Keta lagoon (top), Angor channel connecting Keta lagoon to the Volta River (middle) and in Songor lagoon (bottom)

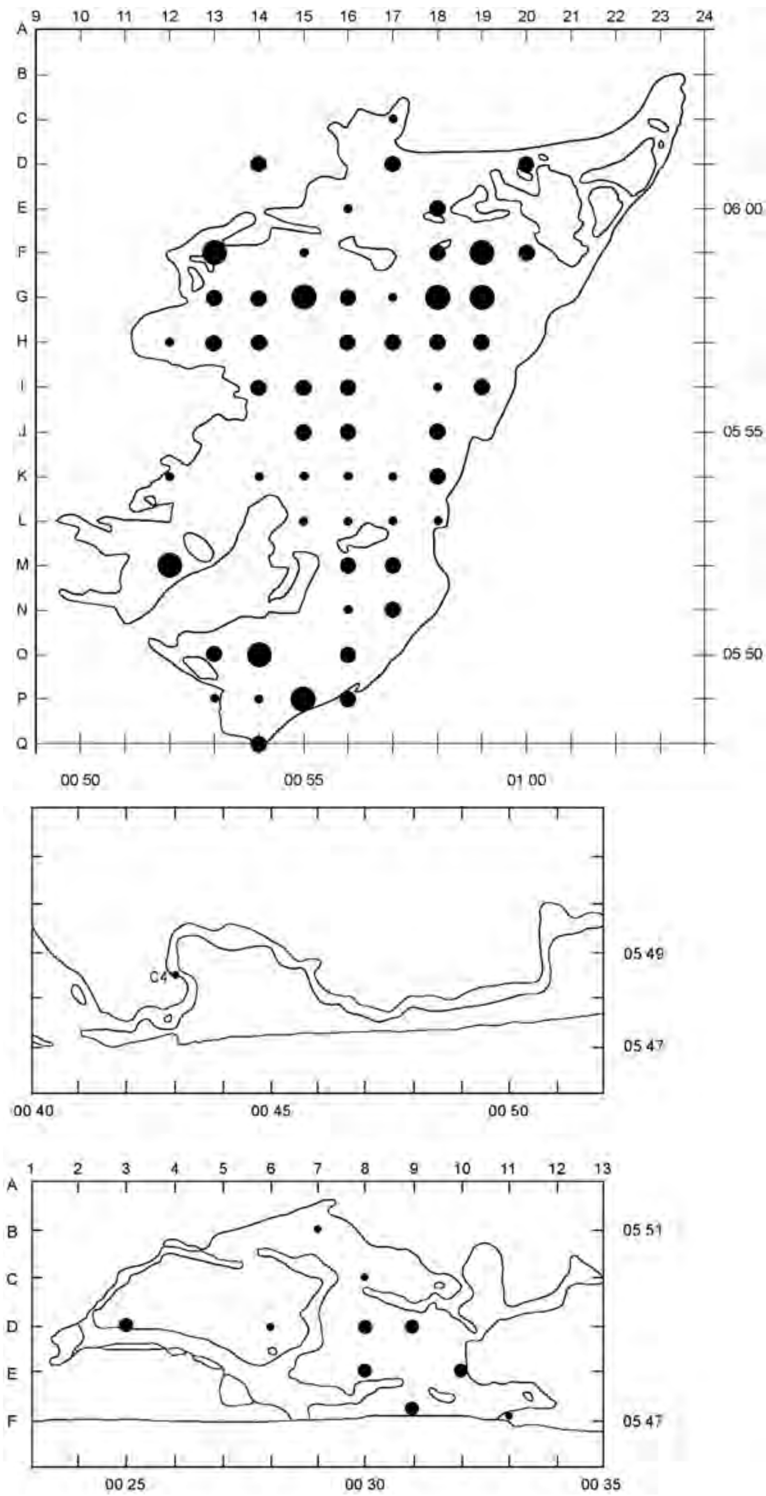

Figure 3 Distribution of *Tympanotonos* in Keta lagoon (top), Angor channel connecting Keta lagoon to the Volta River (middle) and in Songor lagoon (bottom)

The Keta lagoon and the swamps that surround it contained 109 aquatic macrophyte species compared with 57 in Songor and the swamps that surround it. This is likely due to the drier and more saline conditions that occur around the latter. The most dominant aquatic macrophytes are the large emergent species *Typha domingensis, Scirpus littoralis* and the rampant grass *Paspalum vaginatus* with above ground biomass (dry weight) values of 1270 ± 790, 674 ± 358, and 1278 ± 868 g m^{-2} respectively. These species were most common in the freshwater zones around both lagoons (fig 4).

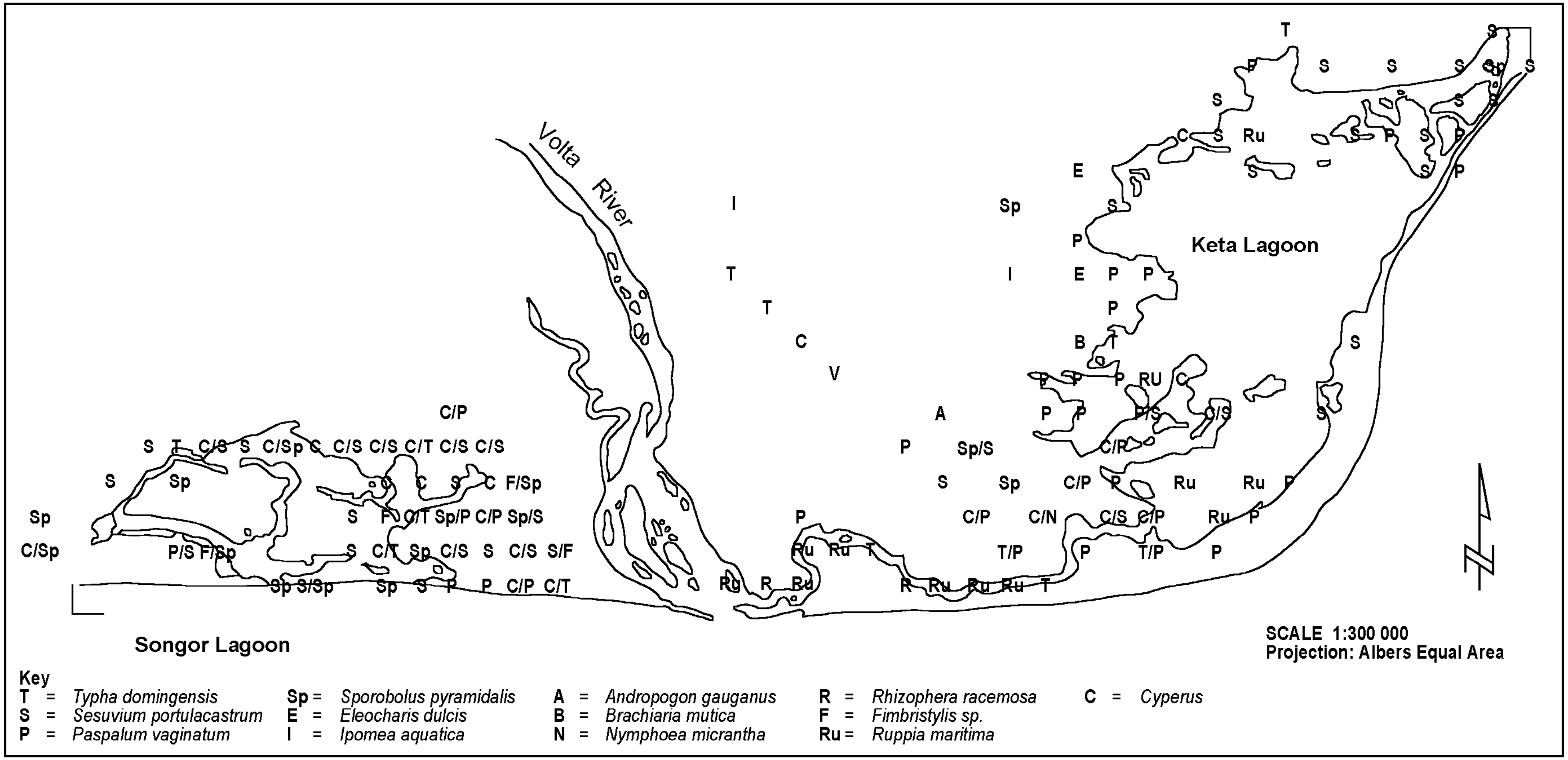

Figure 4 Distribution of aquatic macrophytes around Keta and Songor lagoons

The drier and saline areas around each lagoon are characterised by a *Sesuvium portulacastrum* and *Sporobolus pyriamidialis* association. The diversity of the vegetation around Keta lagoon was shown in transects that generally had *Ruppia maritima* at the waters edge or in the shallow water, followed by *Sesuvium*, then zones dominated by *Paspalum vaginatum, Cyperus articulatus* and/or *Scirpus littoralis,* or *Typha domingensis*. Much of Songor lagoon, especially the western half of the lagoon contained salt ponds and little vegetation.

Management of Keta and Songor lagoons

The description of the ecological character of the lagoons and the development of monitoring programs are components of a management strategy for the long-term sustainable use of the lagoons. The major threats to the lagoons were identified and grouped under four broad categories: water regime; water pollution; physical modification; and exploitation and loss of production as a basis for identifying more specific issues (table 1).

Table 1 Priority pressures in Keta and Songor lagoons

	Wetland	
Priority	**Keta**	**Songor**
1	Erosion and damage from erosion control measures	Expansion of urban infrastructure
2	Flooding and damage from flood control measures	Hunting marine turtles
3	Reclamation of land	Disposal of solid waste-refuse
4	Pollution from sewage	Over-exploitation of fish
5	Pollution from fertilisers	Blockage and diversion of freshwater

As the two lagoons are very large it would be impossible to carry out the same sort of sampling intensity that was used in the baseline study for a monitoring program. A stratified random approach is recommended for monitoring based on further analysis of the pressures occurring in each lagoon. The listing of pressures shown in table 1 does not contain any quantitative data — such information is required before the monitoring projects are implemented. On the whole, the sampling methods described in the report by Finlayson et al (2000) are suitable for further monitoring if supported by an adequate sampling schedule.

It was also recommended that further work be carried out in order to gain an in-depth understanding of the lagoons and how they function, especially in relation to the many goods and services that are derived from the lagoons by local people. It is anticipated that local people would have a wealth of information on such issues and could advise managers and researchers alike. Further, it is recognised that some basic research is required as a base for management decisions, especially those relating to long-term sustainable use of the resources in the lagoons. This is in line with the general goal of gathering information that can be used to ensure the lagoons are used in a sustainable manner.

Acknowledgments

A large number of individuals were involved in this project. Thanks must first go to Mr Bill MacFarlane, Mr HO Ankrah and Mr F Seku for their technical assistance in the field. They were ably supported by staff from the Ghana Wildlife Society and Ghana Wildlife Department. The Ghana National Petroleum Company granted permission to use the Camp at Anloga. And we thank the Ghana Wildlife Department who through the Ghana Coastal Wetlands Management Project invited us to carry out this study.

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Wetland scientists — involvement in training, community awareness and exchange of information

CM Finlayson

Introduction

Across northern Australia local communities have increasingly become involved in wetland management. This has occurred through the development of structured processes, such as those in place in Kakadu National Park (Wellings 1995, Lindner 1999) and less formalised processes, such as those in place in parts of Arnhem Land (Storrs 1999a,b, Szabo 1999). The nature and intent of these processes has varied enormously, generally in response to local needs, resources and opportunity. However, they have all had an emphasis on local participation and many have encouraged interaction and collaboration with scientists and scientific institutions. In unison with the development of more participatory approaches towards wetland management there has been increased involvement of scientists with the general community and across sectoral bounds, resulting in greater exchange of information and knowledge about wetlands.

This has occurred in northern Australia with many individuals and organisations playing a role. In line with this general direction we have developed a number of specific activities designed to increase interaction between wetland scientists and managers and the general community. These have covered general awareness raising within the broader community, specific training of technical experts with input from both scientists and the broader community, and the provision of specific advice on technical issues at local, national and international fora.

The lessons derived from our efforts with members of the wider community interested in wetland management have contributed to many projects including the development of international guidelines for involving local communities and indigenous people in wetland management, community-based monitoring exercises, and specific training programs. Based on these experiences a paradigm for involving local people in a coastal wetland monitoring program was produced (Finlayson & Eliot 2001). An overview of these activities is given below.

At the outset it is emphasised that these efforts were team-based with input from many people, both professional scientists and managers as well as landowners and users. The team effort was seen as a key to both involving members of local communities and increasing interaction between professional scientists and the wider community. It is also noted that these efforts were at times accompanied by many difficulties including those due to the reluctance of some sectors or individuals to participate fully or with equity. Based on information reported from elsewhere (eg see papers in Carbonell et al 2001) it seems that these experiences may not be uncommon.

Community-based monitoring projects

As a component of general outreach activities a number of specific community-based monitoring projects were initiated. Two were undertaken within Kakadu National park: i) monitoring of bird species in the vicinity of an urban lake in the settlement of Jabiru, and ii) monitoring of waterbirds and vegetation in Yellow Water lagoon, a popular tourist and recreational fishing destination. A further project was undertaken at wetlands on a cattle station adjacent to Kakadu National Park.

These projects had dual goals. Foremost was the development of increased awareness by local people of wetlands and their species. This was done through involving local people in the monitoring activities. Within Jabiru this included the involvement of volunteers in regular systematic bird surveys around Jabiru lake. Tour guides were similarly involved at Yellow Water lagoon. While at Carmor station members of the leasees family were involved. Secondly, general data on seasonal changes in the wetlands were collected.

Those involved in these projects were given training in techniques to collect data from specific sites and were at all times supported by qualified scientists and technicians. The results were immediately passed on to the local people through discussion of the data and the major changes that were observed. This led quickly to increased awareness about wetlands and in the case of the tour guides the information was immediately transferred to visitors. Data were collected at regular intervals for 1–2 years and are currently being reviewed and used in formal and informal presentations.

Wetland training

Three formal training courses have been undertaken in collaboration with staff from the Northern Territory University, Darwin. Specifically this included taking responsibility for formal courses on wetland management within a Master of Tropical Environmental Management (www.ntu.edu.au/faculties/science/sbes/pgrad/tropenvman.htm). The courses lasted for 1–2 weeks and were held yearly from 1997–99. They covered general technical subjects such as wetland management and monitoring and included formal lectures, discussions and field work. In addition to input from experienced wetland scientists a concerted effort was made to invite local wetland managers and practitioners to present lectures and engage in discussion with course participants. In July 2000, a further course was presented as a component of the Asia Pacific Wetland Managers Training Program run by NTU and funded by Environment Australia (http://www.ntu.edu.au/ctwm/training.html). This included students specifically invited from a number of countries in Asia and covered similar materials as presented in the Masters courses.

In all courses the participants were presented with current scientific knowledge, such as that on wetland monitoring and assessment being developed through the Ramsar Wetlands Conventions technical panel, introduced to scientific techniques that were either under development or being applied in practical situations, eg aquatic ecotoxicological and stream monitoring techniques, as well as being able to discuss wetland management with on-the-ground managers and owners. Feedback from participants included appreciation at being exposed to real life research and management situations by practitioners rather than the more distant or theoretical treatments that many had experienced on other occasions.

The resource material for such courses came mainly from local studies (eg Bayliss et al 1997, Storrs & Finlayson 1997) and from a collation of papers from local wetland users, owners and managers (Finlayson 1995). Eventually further information on wetland management issues was collated from many local projects and experiences and presented as a compendium for students and managers alike (Finlayson & Spiers 1999a). The emphasis in these courses was on practical issues as opposed to theoretical analyses or textbook recitals.

Study tours and specific training events

In addition to the formal training course mentioned above a number of informal training courses were held. These included visitors from Tasek Bera, Malaysia, and the Mekong Delta, Viet Nam. The subject material was similar to that presented in the training course with the main difference being that the participants were given a greater opportunity to develop a program of specific interest to themselves. These tours have resulted in ongoing contact and communication and the development of joint project activities (eg *Vulnerability assessment of major wetlands in the Asia-Pacific region to climate change and sea level rise* — van Dam et al 1999b; and the Asian Wetland Inventory — Finlayson et al 2002).

Specific training and interaction has also occurred with local aboriginal communities. This has involved employment and training of individuals either in specific sampling projects, eg fish sampling of lowland billabongs, or more general field station duties. Input has also been given to special workshops and meetings, such as those on wise use of wetlands by Aboriginal people (Centre for Indigenous Natural and Cultural Resource Management 1997, Whitehead et al 1999) as well as many informal meetings and discussions about wetland management issues. Specific interaction has also been encouraged with technical staff from other institutions and countries (eg Czech Republic, Ghana, Papua New Guinea, The Netherlands, Switzerland and the United Kingdom) and visits by community groups or their representatives. The latter has been extended to occasional hosting of specialist environmentally-oriented tour groups and linked with celebration of World Wetlands Day on 2 February (in celebration of the formulation of the text of the Wetlands Convention in Ramsar, Iran, 1971). Regular effort is also given towards presentations and displays at local fetes, open days and other local celebratory or awareness raising events hosted by other organisations. Many of these have included wider representation of the research activities undertaken at ***eriss***.

Hosting and/or participation in such events has occurred through both structured and opportunistic processes as appropriate. This dual approach has had many advantages and has enabled scientific staff to develop specific interests and impart knowledge as well as respond to informal situations and both contribute to and learn from different forms of interaction. In this manner there has been a greater sharing of knowledge and mutual learning than may otherwise have occurred if formal scientific conferences and the like were the only focus.

Local community wetland management

At the invitation of landholders and interested agencies assistance and advice on wetland management issues has been provided through a number of formal processes and activities. The nature of these activities has varied and in part reflects the nature of the differing juridictional structures in the region. Finlayson et al (1998) have compared in brief the arrangements within Kakadu National Park and adjacent land and noted that the involvement of local people in wetland management raises a number of challenges:

- Community consultation takes time and potentially causes delays and challenges established authority lines.
- It can be difficult to establish the representativeness of views being promulgated.

- A common understanding of fundamental issues may be elusive.
- Local interests may conflict with broader interests.

They also note that 'governmental resources allocated for wetland management are not evenly disributed between particular issues and across locations'.

Within the Alligator Rivers Region, which includes Kakadu National Park, several approaches have been adopted towards assisting local communities with wetland management. One approach has involved catalysing interest from multiple stakeholders in a coastal monitoring program. Finlayson and Eliot (1999) have described the basis of this program and proposed a coastal monitoring node to facilitate ongoing assessment of the coast, in particular the wetlands, to the effects of short-term changes in climate and other environmental factors that occur within planning horizons of approximately 100 years. Eliot et al (2000) outline the major components of the program under the following headings:

- Introduction to the coastal monitoring program
- Data collection and information management
- Collation of baseline information: regional processes
- Collation of baseline information: the coastal plain
- Application and accountability

As consultation with multile stakeholders was a major component of the program Finlayson and Eliot (2001) also described the manner in which this was undertaken (see below).

Wetland management issues have also addressed and assistance and advice provided at the behest of local people and/or the park service in response to their needs for further scientific data (Wellings 1995, Lindner 1995, 1999). This resulted in specific investigations, such as the extent of hydrocarbon pollution in Yellow Water lagoon (van Dam et al 1999) and at Gunlom waterfall from chemicals used by visitors for personal protection from the sun and insects (Rippon et al 1994). On both ocassions adverse results were not found, but it was stressed that further and more detailed investigations could be warranted. A further analysis involved investigating the effect of herbicides used to spray the floating weed *Salvinia molesta* (Finlayson et al 1994a). Again, data and recommendations were provided to the park managers. Involvement in weed management built on a longer term involvement with weed issues in Kakadu (Cowie et al 1988, Finlayson et al 1994b), including provision of an analysis of the spread of paragrass (*Brachiaria mutica*) within the Magela Creek wetlands from being sparsely distributed in the mid 1980s (Finlayson et al 1989) increasing to 920 ha coverage in 1996 (Knerr 1998).

To the west of Kakadu local landholders had formed a land care group that became known as the Mary River Landcare group. Expert opinion on issues as diverse as climate change and wetlands (Bayliss et al 1997), weed management and saline intrusion was provided during regular meetings of this group. Formal input to the development of a Total Catchment Management Plan for the Mary River was made through involvement in the landcare group. The concept of wise use was also raised in a paper (Finlayson 1996a) presented at a workshop to address land use options for the wetlands in the lower Mary (Jonauskas 1996). This paper raised the spectre that whilst wise use was an attractive concept (akin to sustainable development) it may be elusive unless supported by ongoing commitment and assessment of options and outcomes.

A major effort was also undertaken in the Blythe-Liverpool wetlands in Arnhem Land to the east of Kakadu. After being approached by representatives of local landholders and after further consultation a series of wetland and river surveys were conducted (Finlayson et al 1997). These included working with local Aboriginal people to sample fish, macro-invertebrates and vegetation at a number of locations (Pidgeon & Boyden 1997, Thurtell 1997). The results were reported and on ocassions formal training was provided to members of the community involved. Through the formally established Top End Indigenous People's Wetland Program (Storrs 1999a,b) a review of information sources that could be useful for developing a management plan for the Blythe-Liverpool wetlands was undertaken (Thurtell et al 1999). This review drew heavily on information available from published sources and from dialogue with local people. It was also accompanied by a brief commentary on the Ramsar Wetlands Convention and its key principles of wise use and involvement of local communities and indigenous people in wetland management. The latter were developed by a cooperative effort based on case studies drawn from around the world, including the Blythe-Liverpool wetlands (Hunziker et al 1999). Representatives from the local community were interested in further information on the Ramsar Convention. This was achieved through direct contact with representatives of the Convention and its subsidiary scientific and technical panel and other interested parties. This dialogue has continued (eg Finlayson 1999a) with a view to enabling local people to determine their own interest and possible direct participation in this international Convention.

Problems for wetland managers of the future were also addressed (Finlayson et al 1997, Finlayson & Pidgeon 1999). In particular it was stressed that indigenous wetland managers and owners in northern Australia may need to address: allocation of environmental flows; development of acid sulphate soils; adaptation to climate change and sea level rise; control and management of feral animals and weeds; and responses to proposals to construct tidal power barrages. In raising awareness of these issues Finlayson & Pidgeon (1999) noted the following 'We would like our scientific expertise to assist them [people] in making wise choices and to prevent the potential disasters that could befall the valued wetlands of northern Australia.' This statement is as true today as it was when the talk that formed the basis of the paper by Finlayson and Pidgeon (1999) was presented in 1998 at a workshop on the Wise Use of Wetlands in Northern Australia (Whitehead et al 1999).

Involving local communities in an assessment and monitoring program

In developing a coastal wetland assessment and monitoring program for the Alligator Rivers Region (see Eliot et al 1999a) a paradigm for involving local communities in assessment and monitoring activites was developed (Finlayson & Eliot 2001). The paradigm revolves around formal consultation involving interested and relevant community groups and governmental agencies coupled with scientific rigour and feedback to participants. It includes the following steps:

- Establishment and empowerment of an expert assessment and monitoring centre – based on discussion with key stakeholders and recognition of all technical competencies.
- Consultation with and empowerment of key stakeholders, including the local community – identification of key stakeholders and their individual roles at the start of the process and supported by formal and informal meeting processes to both develop awareness and seek advice and assistance where practicable.

- Identification of major processes and causes of ecological change – primarily a technical exercise but honed through discussion with and input from local residents etc.
- Collation and coordination of available data and information – involving rigid data management protocols to enhance access and store/file information.
- Identification of potential collaborators and partners – an ongoing and iterative process involving technical and lobby groups.
- Design and implementation of technical assessment and monitoring projects – a technical task based on the best available knowledge and advice from as many sources as feasibly possible.
- Audit and, if necessary, termination of assessment and monitoring projects – a review process that involves outside advice and participation.
- Implementation of management prescriptions based on results of the assessment monitoring projects — dependent on the establishment and maintenance of links with management agencies and officials.
- Provision of feedback to stakeholders, partners and community groups — an ongoing and iterative process whereby awareness and trust are established and maintained.
- Audit of management outcomes and readjustment of the monitoring program in the context of impacts arising from the management strategies adopted — a process to ensure that the best available information was being used for management purposes in an adaptive manner.

The relative merit of each step of the paradigm is dependent on local circumstances, such as the interest of the local community groups and their interaction with governmental officials. Whilst the emphasis on individual steps may vary it is extremely unlikely that any step will be completely bypassed without placing the entire process in jeopardy.

Formal advice and presentation of scientific information

The knowledge and experience developed from research activities undertaken in the tropical wetlands of northern Australia, particularly those within Kakadu National Park, have been used to provide guidance and assist wetland managers and policy makers elsewhere. The provision of advice to Australian national programs on the National River Health Program (AUSRIVAS component) (Environment Australia), and the Revision of the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (a publication of the National Water Quality Management Strategy) (for ANZECC and ARMCANZ and coordinated by Environment Australia) has been described elsewhere in this volume ('AUSRIVAS operator sample processing errors: Implications for model sensitivity', Humphrey; 'An overview of the new water quality guidelines for Australian and New Zealand aquatic ecosystems', Humphrey & McAlpine).

Other activities have included input into the Porgera Environmental Advisory Komiti in Papua New Guinea, a mining project that appointed an independent committee to assess its environmental management (Placer Dome Asia Pacific 2000). Independent scientific advice has also been proved on the water reform processes in NSW specifically through the Macquarie-Cudgegong River Management Committee in New South Wales (Finlayson 2001). These activities have seen the direct transferrance of knowledge and expertise to wetland managers and owners and a greater understanding by scientists of the issues involved in community-based environmental committees.

Scientific knowledge developed in the region has also been used to support the efforts of the Australian Society for Limnology to outline key issues for inland waters in Australia and present a list of major challenges facing aquatic and wetland ecosystems (www.asl.org.au/asl_poldoc_challenges.htm). The list includes: provision of surface and groundwater for environmental benefits; prevention of pollution and contamination of aquatic habitats; prevention and reduction of salinisation of wetlands; prevention of further drainage and infilling of wetlands; management of grazing in wetlands; restoration and protection of riparian vegetation; prevention and control of invasive species; mitigation of climate change and sea level rise; and development of rigorous inventory, assessment and monitoring protocols.

Support was also given to the alliance of the Australian Society for Limnology, Wetland Care Australia and WWF-Australia that formed the Australian Wetland Forum to produce A Strategy to Stop and Reverse the Loss and Degradation of Australian Wetlands (www.asl.org.au/asl_wetlandforum.htm). This outlined strataegies and actions that should be taken to achieve the goal of stopping and reversing the loss and degradation of Australian wetlands. The strategy was divided into four components: telling stories about wetlands; implementing policies and initiatives; providing mechanisms to involve all sectors; and a view of the next steps.

Scientific knowledge has also been used to develop national and international protocols for wetland inventory (Finlayson 1999b, Finlayson et al 1999), risk assessment (van Dam et al 1999a) and monitoring (Finlayson 1996b,c). Contributions have also been made to the Intergovernmental Panel on Climate Change third assessment report (Gitay et al 2001, Pittock & Wratt et al 2001). Such efforts continue and have seen the involvement of international wetland scientists in the program of work being conducted in northern Australia. This adds a further dimension to the expertise and knowledge available for managing wetlands in northern Australia and complements the efforts being made to involve local people and other stakeholders and to share knowledge and experience. Technical issues covered have included the reasons for the loss and degradration of Australian wetlands (Finlayson & Rea 1999) and the status of wetland inventory in Australia (Finlayson & Spiers 1999b) and globally (Finlayson & van der Valk 1995, Finlayson & Spiers 1999c).

The National Centre for Tropical Wetland Research

The communication, consultation and training aspects of the wetland research program undertaken have been formally included in the working process of the National Centre for Tropical Wetland Research (***nctwr***). The ***nctwr*** is a formal alliance between four research and/or training institutions: James Cook Univeristy, Northern Territory University, the University of Western Australia and the Environmental Research Institute of the Supervising Scientist. It is governed by a formal agreement that establishes the aim to 'promote the wise use of tropical wetlands' through research and training programs with an emphasis on efective consultation with stakeholders. Training directions encompass formal and academic programs (with formal accreditation) as well as informal and field-based training, with specific short courses for researchers, managers, owners and users of tropical wetlands. The research directions include: assessing existing and potential threats to wetlands; devleoping procedures and standards for monitoring wetlands; developing procedures and standards to sustainably use and restore wetlands; investigating physical, chemical and biological processes in wetlands; and describing the values and benefits derived from wetlands.

The ***nctwr*** incorporates a formally constituted board with representatives from each partner and an independent chair. It also incorporates a broad stakeholder group known as the Advisory Committee. Membership of this committee currently includes national/international non-governmental organisations, federal and provincial governmental agencies, industry representatives and community based organisations. Membership is regularly reassessed based on the need for advice on wetland research and training priorities. The committee is specifically asked to provide advice and input and to support project based activities undertaken by staff from the partner organisations and other collaborators.

The development of the ***nctwr*** is seen as one outcome of many past collaborative efforts involving scientists and wetland managers in northern Australia. It is also seen as a vehicle for promoting further collaboration and for maximising efforts to support wetland conservation and management of tropical wetlands.

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WETLAND ECOLOGY AND CONSERVATION

Risk Identification and Assessment

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

RA van Dam[2], C Camilleri, CJ Turley & SJ Markich[3]

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.

[2] Formerly ***eriss***. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.

[3] Environment Division, ANSTO, PMB1, Menai, NSW, 2234, Australia

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.

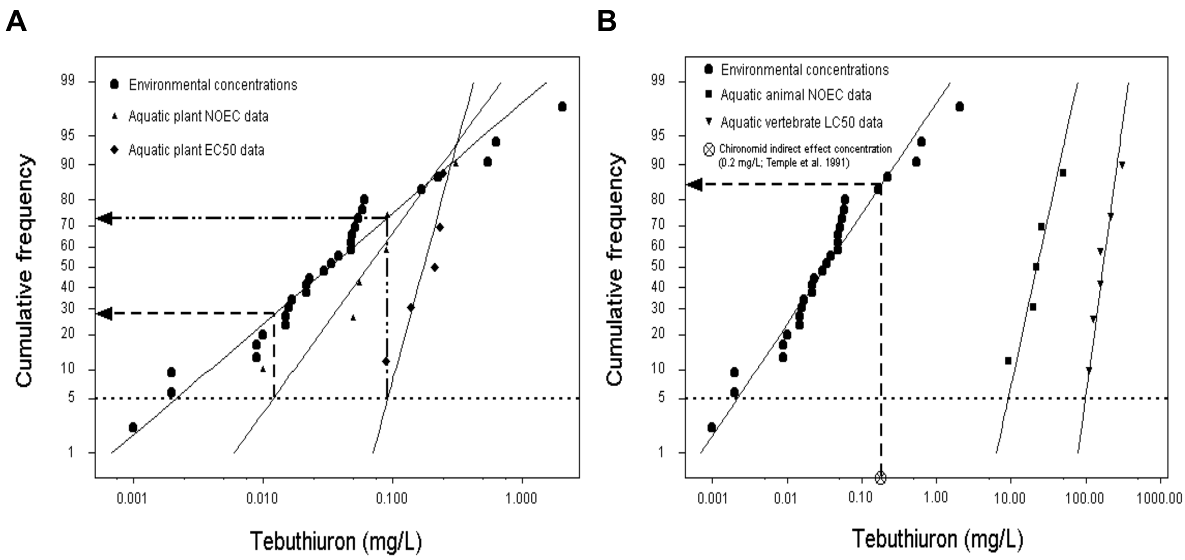

Figure 1 Comparison between the distribution of environmental tebuthiuron concentrations and (A) chronic plant sensitivity distributions for tebuthiuron based on NOEC data and EC_{50} data, and (B) chronic animal sensitivity and acute vertebrate sensitivity distributions for tebuthiuron based on NOEC 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).

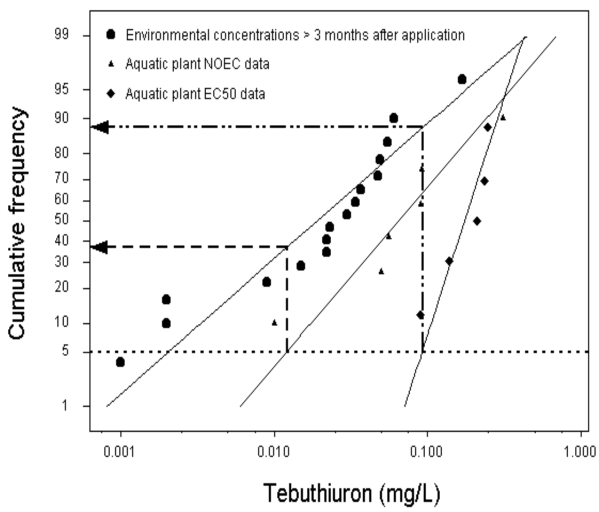

Figure 2 Comparison between the distribution of environmental tebuthiuron concentrations measured ≥3 months following application and the chronic plant sensitivity distributions for tebuthiuron based on NOEC data and EC_{50} data

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 outweigh 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 outweighed 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.

2 Formerly ***eriss***. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.

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.

2 Formerly ***eriss***. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.

3 Department of Environment and Natural Resources, Region 7, Banilad, Mandaue City, Cebu, Philippines, 6014.

4 First Institute of Oceanography, State Oceanic Administration, Qingdao, China, 266003.

5 Wetlands International-Oceania, GPO Box 787, Canberra, ACT, 2601, Australia.

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.

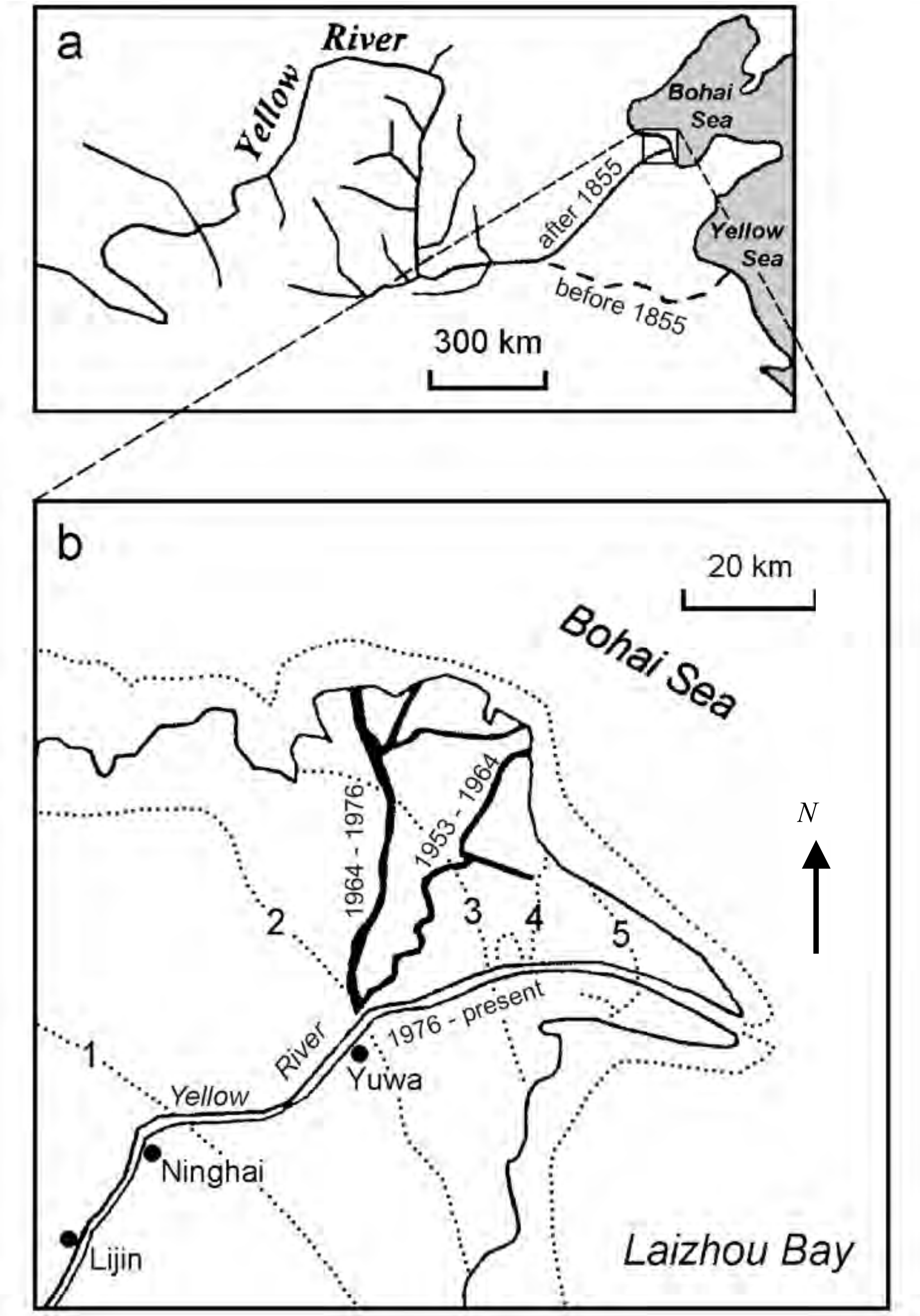

Figure 1 Map of (a) the location of the Yellow River basin, and (b) the modern Yellow River delta with the historical changes of the coastline: 1. coastline of 6000 years BP; 2. coastline of 1855; 3. coastline of 1934; 4. coastline of 1976; 5. coastline of 1980

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.

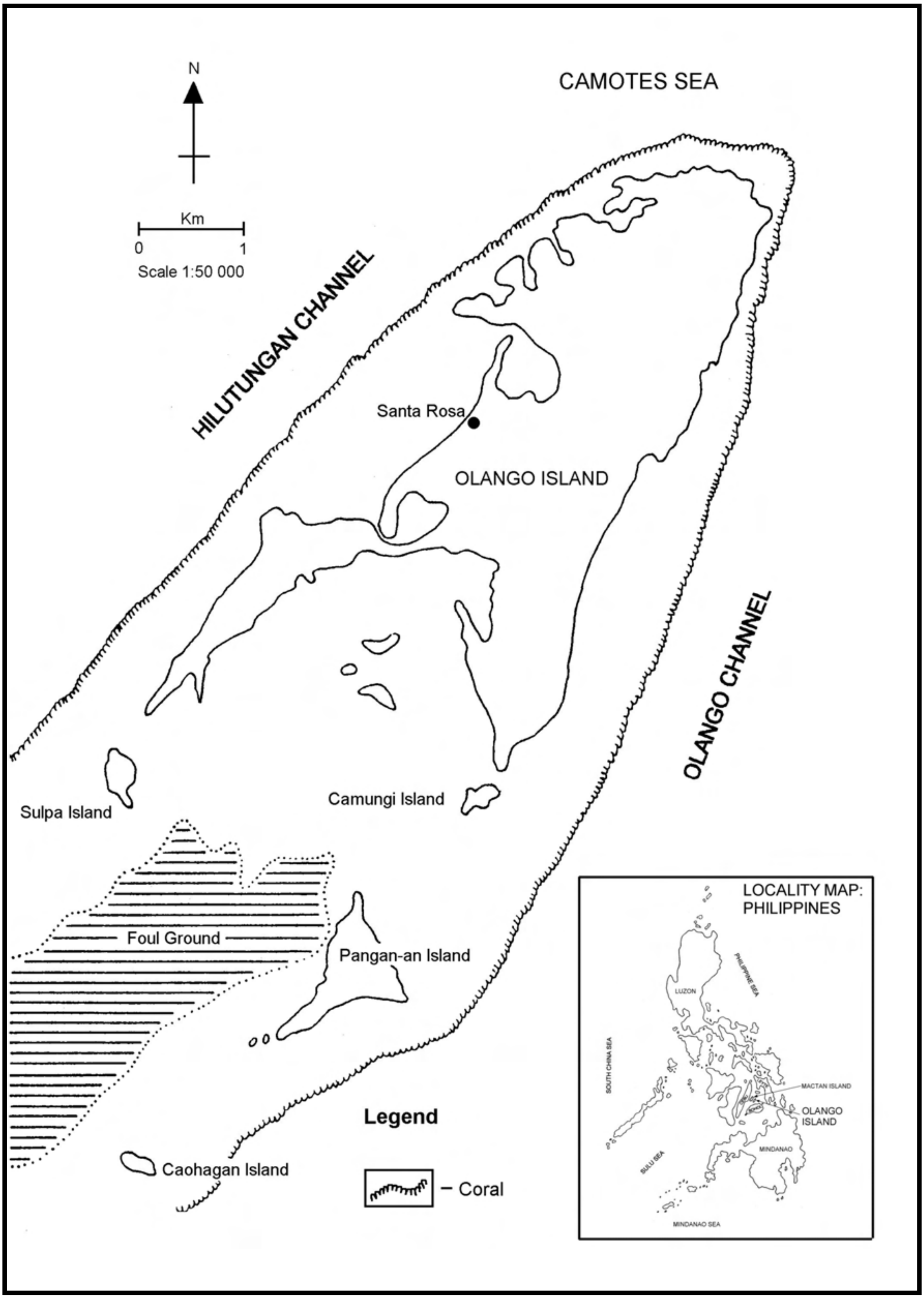

Figure 2 Map of Olango Island showing the major geographical features

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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1 Formerly *eriss*. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.

2 Northern Land Council, Darwin, PO Box 42921, Casuarina NT, Australia.

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').

2 Formerly ***eriss***. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia

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

Observed NOECs	Predicted 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}*

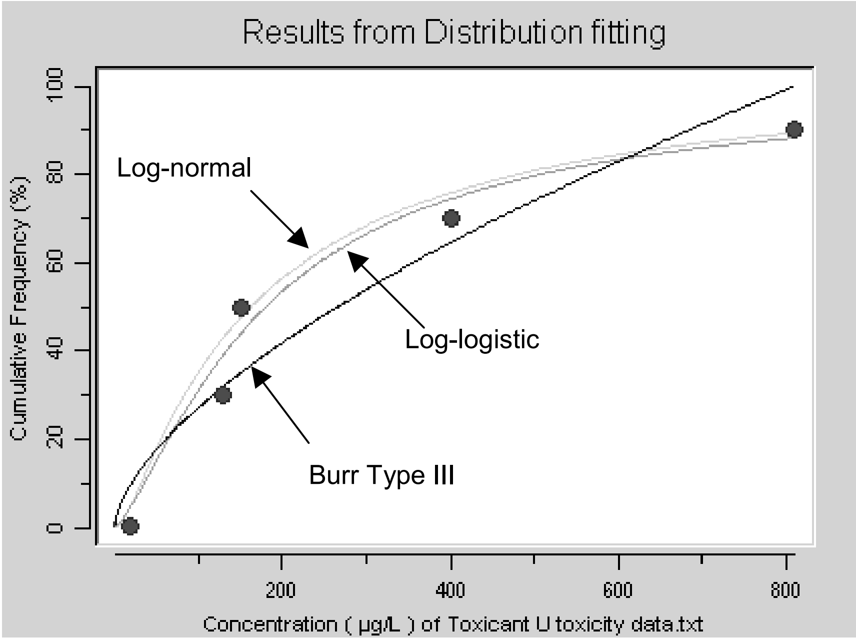

Figure 1 Graphical output of BurrliOz curve-fitting to uranium NOEC values for local species

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.

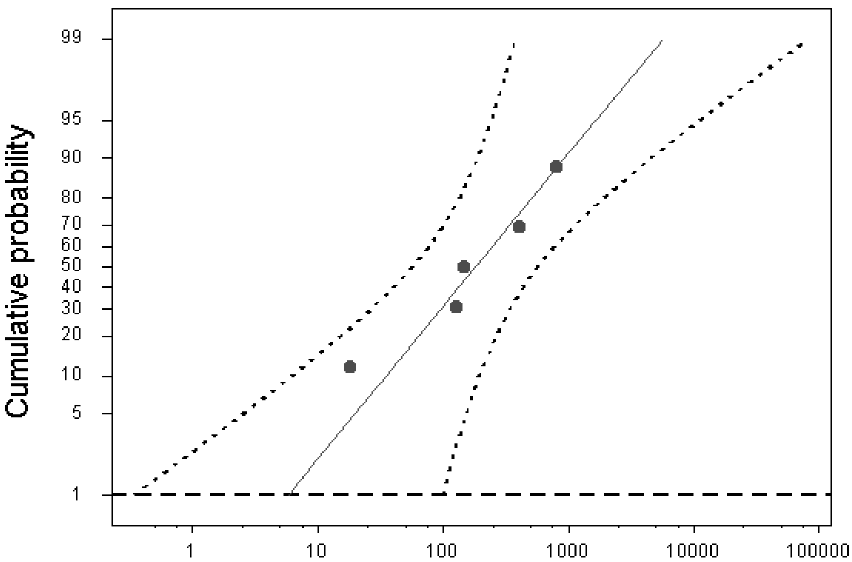

Figure 2 Log-logistic distribution fitted to uranium NOEC values for local species. Dotted lines represent 95% confidence limits.

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').

2 Environment Division, Australian Nuclear Science and Technology Organisation, Private Mail Bag 1, Menai, New South Wales 2234, Australia.

3 National Research Centre for Environmental Toxicology, PO Box 84, Archerfield, Queensland 4108, Australia.

4 Mines Division, Northern Territory Department of Mines & Energy, GPO Box 2901, Darwin, NT 0801, Australia.

5 Formerly *eriss*. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.

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 ($\mu S\ cm^{-1}$)	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 ($\mu g\ L^{-1}$)	89	156	87
Dissolved Al ($\mu g\ L^{-1}$)	33	137	16
Labile Al ($\mu g\ L^{-1}$)	28	118	3.8
Total Mn ($\mu g\ L^{-1}$)	67	34	10
Filtered Mn ($\mu g\ L^{-1}$)	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.

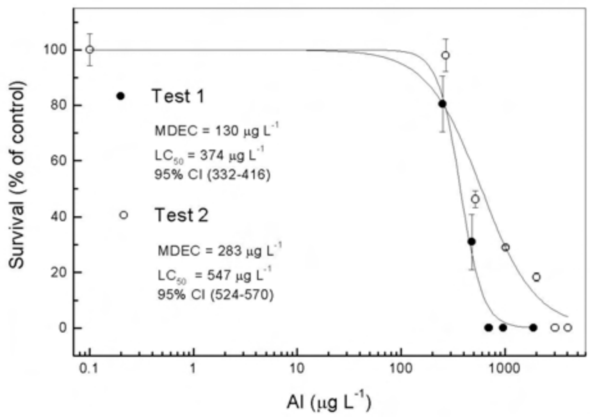

Figure 1 Concentration-response relationships for survival of *M. mogurnda* sac fry exposed to Al in laboratory water at pH 5.0. Data points represent the mean ± 95% confidence intervals. MDEC, minimum detectable effect concentration.

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)

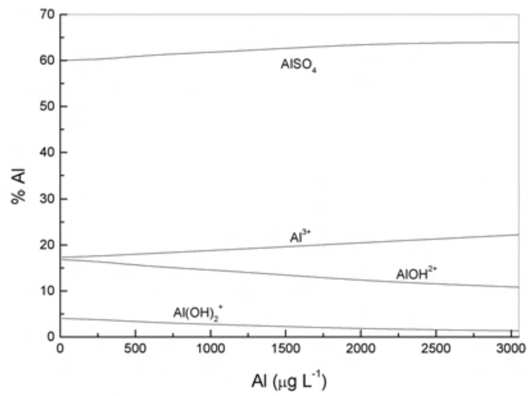

Figure 2 Predicted speciation (% distribution) of Al in laboratory water at pH 5.0

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).

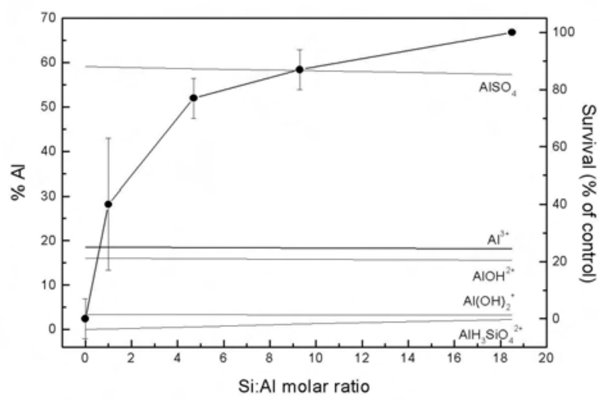

Figure 3 Predicted speciation (% distribution) of Al, together with the concentration-response relationship for survival of *M. mogurnda* sac fry, in laboratory water at pH 5.0 with an increasing Si:Al ratio (Test 4)

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.

Acknowledgments

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Appendices

Appendix 1 List of external authors and contact information

Author	Contact information
Boggs GS	GIS, Faculty of Science & Education, Northern Territory University, Darwin NT 0909. Present address: Department of Geography, University of Western Australia, Crawley WA 6009.
Davidson NC	Wetlands International, Wageningen, The Netherlands. Present address: Ramsar Convention Bureau, Gland, Switzerland.
Devonport C	GIS, Faculty of Science & Education, Northern Territory University, Darwin NT 0909 (chris.devonport@ntu.edu.au)
Eliot I	Department of Geography, The University of Western Australia, Perth, Western Australia, Australia (iane@sunny.gis.uwa.edu.au)
Erskine WD	Environmental Management & Forest Practices Directorate, State Forests of NSW, Locked Bag 23, Pennant Hills, NSW 2120 (now Supervising Scientist Division).
Gordon C	Volta Basin Research Project & Zoology Department, University of Ghana, Legon, Accra, Ghana.
Grabham M	School of Geosciences, The University of Newcastle, Callaghan, NSW 2308. Present address: NSW Agriculture, PO Box 1087, Griffith, NSW 2680.
Hancock GR	School of Geoscience, The University of Newcastle, Callaghan, NSW 2308
Mapalo A	Department of Environment and Natural Resources, Region 7, Banilad, Mandaue City, Cebu, Philippines, 6014
Markich SJ	Environment Division, Australian Nuclear Science and Technology Organisation, Private Mail Bag 1, Menai, New South Wales 2234, Australia.
McAlpine KW	Department of Environmental Protection, PO Box K822, Perth WA 6842
Noller BN	Mines Division, Northern Territory Department of Mines & Energy, GPO Box 2901, Darwin, Northern Territory 0801, Australia.
Ntiamoa-Baidu Y	Zoology Department, University of Ghana, Legon, Accra, Ghana; Ghana Wildlife Society, Accra, Ghana. Present address: WWF International, Avenue du Mont-Blanc, 1196 Gland, Switzerland.
Parker G	Mines Division, Northern Territory Department of Mines & Energy, GPO Box 2901, Darwin, Northern Territory 0801, Australia.

Peiying L	First Institute of Oceanography, State Oceanic Administration, Qingdao, China, 266003
Stevenson NJ	Wetlands International, Wageningen, The Netherlands. Present address: Ecoscope, Cambridge, United Kingdom.
Storey AW	Current address: Department of Zoology, The University of Western Australia, Nedlands, WA 6907
Storm J	Department of Physics, University of Adelaide, Adelaide 5005 South Australia
Storrs M	Northern Land Council, PO Box 42921, Casuarina, NT 0811, Australia
Thurtell L	Formerly Supervising Scientist Division. NSW Department Land and Water Conservation, PO Box 136, Forbes, NSW 2871
van Dam R	Formerly Supervising Scientist Division. Current address: Sinclair Knight Merz Ecotoxicology Laboratory, PO Box 164, St Leonards, New South Wales 1590, Australia.
Waterman P	Environmental Management Services, Canberra, Australia (waterman@orac.net.au)
Watkins D	Wetlands International-Oceania, GPO Box 787, Canberra, ACT, 2601, Australia
Willgoose Garry	Dept Civil, Surveying and Environmental Engineering, The University of Newcastle, University Drive, Callaghan, NSW 2308.

Appendix 2
External publications by *eriss* staff and consultants to the Supervising Scientist during 1995–2000

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Appendix 3 Supervising Scientist Research publications[1]

Supervising Scientist Research Summaries

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SSR101 Finlayson CM (ed) 1995. *Wetland research in the wet-dry tropics of Australia*. Workshop, Jabiru NT 22–24 March 1995, Supervising Scientist Report 101, Supervising Scientist, Canberra. (293 pp)

SSR102 Beer Tom & Ziolkowski Frank 1995. *Environmental risk assessment: An Australian perspective*. Supervising Scientist Report 102, Supervising Scientist, Canberra. (138 pp) [Available only as HTML files by email]

SSR103 Johnston Arthur & Martin Paul 1995. *Rapid analysis of 226Ra in mine waters by gamma-ray spectrometry*. Supervising Scientist Report 103, Supervising Scientist, Canberra. (26 pp)

SSR104 McQuade Christopher V, Johnston John F & Innes Shelley M 1995. *Review of historical literature and data on the sources and quality of effluent from the Mount Lyell lease site*. Mount Lyell Remediation Research and Demonstration Program. Supervising Scientist Report 104, Supervising Scientist, Canberra. (86 pp)

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SSR144 Finlayson CM & Spiers AG (eds) 1999. *Global review of wetland resources and priorities for wetland inventory*. Supervising Scientist Report 144, Supervising Scientist, Canberra. (also available on CD). (524 pp)

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SSR165 Moliere DR, Evans KG, Willgoose GR & Saynor MJ 2002. *Temporal trends in erosion and hydrology for a post-mining landform at Ranger Mine*. Northern Territory. Supervising Scientist Report 165, Supervising Scientist, Darwin NT. (128 pp)

SSR166 Rovis-Hermann J, Evans KG, Webb AL & Pidgeon RWJ (eds) 2002. *Environmental Research Institute of the Supervising Scientist research summary 1995–2000*. Supervising Scientist Report 166, Supervising Scientist, Darwin NT. (212 pp)

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SSR170 Supervising Scientist 2002. *Investigation of the stockpiling and reporting incidents at Ranger and Jabiluka 2002*. Supervising Scientist Report 170, Supervising Scientist, Darwin NT. (58 pp)

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RR2 Hart BT & McGregor RJ 1982. *Water quality characteristics of eight billabongs in the Magela Creek catchment*. Research report 2, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra. (60 pp)

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RR9 Woodland DJ & Ward PJ 1992. *Fish communities in sandy pools of Magela Creek, Alligator Rivers Region*. Research report 9, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra. (88 pp)

RR10 Willett IR, Bond WJ, Akber RA, Lynch DJ & Campbell GD 1993. *The fate of water and solutes following irrigation with retention pond water at Ranger Uranium Mine.* Research report 10, Supervising Scientist for the Alligator Rivers Region, AGPS, Canberra. (132 pp)

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