

Contamination of Australian Groundwater Systems with Nitrate

Occasional Paper 03/99

SINCLAIR KNIGHT MERZ



**Land & Water
Resources**
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Contents

Executive summary	1
Introduction	5
Study organisation	5
Study aims	5
Nitrate contamination—background	6
Nitrate contamination in groundwater—worldwide	6
Introduction of nitrate to aquifers	6
Nitrate contamination in Australia	9
Beneficial use and acceptable nitrogen concentrations	9
Distribution of elevated nitrate concentrations in Australia	9
Problems arising from nitrate contamination	10
Case studies	14
Approach to case studies	14
Selection of study areas	14
Case studies	16
Monitoring of nitrate in Australia	28
The nature of the nitrate concentration data	28
Nitrate analytical data	29
Significance of nitrate contamination	32
Impacts of nitrate contamination on groundwater beneficial use	32
Characteristics of nitrate sources and impacts	33
Aquifer conditions and observed nitrate concentrations	37
Observed nitrate concentrations for different nitrate sources	37

Potential for future nitrate contamination	39
Continuing sources of nitrate to aquifers	39
Possible future directions for nitrate releases and loads for various land uses in Australia	41
Management of nitrate contamination in groundwater	43
Overall understanding of the nitrate problem	43
Technical issues required for management	43
Policy implications	45
Conclusions	47
Nature of the problem	47
Data limitations	47
Nature and extent of nitrate sources	47
Potential future impacts	48
Understanding of risks and processes	48
Recommendations	49
Management and policy development	49
Confirmation of nitrate trends	49
Research activities	49
Funding allocations	50
References	51
Case Studies	
1. Pasture, mixed agriculture and forestry—south-east of South Australia	54
2. Effluent disposal—Western Treatment Plant, Werribee, Victoria	70
3a. Septic tank study—Nepean Peninsula, Victoria	79
3b. Septic tank study—Venus Bay and Sandy Point, Victoria	86
3c. Septic tank study—Benalla, Victoria	88
4. Natural process nitrate contamination—Yulara, Northern Territory	91
5. Urban land use and mixed agricultural—Perth Metropolitan and Jandakot Mound	94
6. Urban and rural residential—Wagga Wagga	121
7. Intensive agriculture—horticulture and cropping—Bundaberg	123
8. Mixed agricultural land use—Peel Valley, NSW	129
9. Country town, rural cropping—Narromine, NSW	132

List of figures and tables

Figures

Figure 1		Figure 13	
Summary of key nitrate		Location of Nepean Peninsula	
generation processes	7	groundwater observation bores	81
Figure 2		Figure 14	
Distribution of bores with		Benalla septic tank study area	89
nitrate analyses	11	Figure 15	
Figure 3		Nitrate concentrations in the upper	
Fluctuations of nitrate-N with time in		and lower aquifer	89
groundwater at Jandakot, WA	30	Figure 16	
Figure 4		Location of the Perth groundwater	
Nitrate concentration (mg/L) versus depth in		priority protection areas	97
southwest Victorian Aquifers showing high		Figure 17	
concentrations more common at shallow		Increase in nitrate-N concentrations with	
(<30 m) depth, but significant numbers at		time in two production bores downgradient	
greater depths	38	of unsewered residential area (after	
Figure 5		Barber <i>et al.</i> , 1996)	108
Diagrammatic representation		Figure 18	
of nitrate concentrations in unsaturated		Trends in nitrate concentration in	
and saturated zones	40	groundwater for 1997 to 1994 on the	
Figure 6		Jandakot mound	111
Land use for Mt. Gambier study area	55	Figure 19	
Figure 7		Trends in nitrate concentration in	
Nitrate concentration at		groundwater for 1993 to 1991 on the	
various bore depths	58	Jandakot mound	113
Figure 8		Figure 20	
Nitrogen flows and balance in irrigated		Location of monitoring bores on the	
and non-irrigated paddocks	61	Jandakot mound which had continuous	
Figure 9		groundwater quality data	115
Zonal chemical structure of a nitrogen		Figure 21	
plume at Yahl	64	Concentrations of nitrate-N in groundwater	
Figure 10		west of the Jandakot UWPCA fluctuate over	
Projected nitrate concentration profile		the years and months.	116
histories	65	Figure 22	
Figure 11		Concentrations of nitrate-N in groundwater	
Locality of the Western Treatment Plant		west of the Jandakot UWPCA fluctuate over	
study site	71	the years and months	117
Figure 12		Figure 23	
Layout of the Delta trial site	73	Trends in nitrate concentration—	
		water supply bores, Bundaberg	126

Figure 24	
Rainfall (a), nitrate concentration (ppm as NO ₃), and depth to standing water for 16 bores (after CSIRO, 1996) . . .	127
Figure 25	
Distribution of nitrate—Peel Valley, NSW	131
Figure 26	
Plan of the Narromine Town Area	134
Figure 27	
Conceptual model based on nitrogen species movement at Narromine	134

Tables

Table 1	
Responsibilities and priorities for future works	4
Table 2	
Acceptable nitrogen concentrations for beneficial use	10
Table 3	
Study areas selected for detailed investigation	16
Table 4	
Characterisation of nitrate sources and impacts	34
Table 5	
Responsibilities and priorities for future works	50
Table 6	
History of studies of recharge and nitrate concentrations in groundwater in the south-east of South Australia	54
Table 7	
Percent variation of nitrate-N concentrations explained by different variables, determined by regression analysis	59
Table 8	
Land use and estimated nitrogen loads to study aquifer area	66
Table 9	
Raw effluent nitrogen concentrations	76
Table 10	
Nitrogen balance at Delta trial site	77
Table 11	
Chemical and microbiological groundwater analyses	80
Table 12	
Nitrite and nitrate (NO _x) concentrations in groundwater samples	80
Table 13	
Nitrate/nitrite in groundwater samples from Venus Bay and Sandy Point (January 1994–April 1996)	87
Table 14	
Nitrate concentrations in soil profiles	92
Table 15	
Information on nitrate concentrations from the Bundaberg area (from Keating <i>et al.</i> , 1996)	124

Executive summary

An investigation to evaluate the significance of nitrate contamination in groundwater in Australia has been undertaken on behalf the Land and Water Resources Research and Development Corporation (LWRRDC).

Scope of the study

The overall objective of the study was to provide an overview of the current conditions of groundwater systems in Australia with respect to nitrate contamination and to evaluate the potential for ongoing problems of nitrate contamination.

A series of case studies was conducted to evaluate the current status of nitrate contamination in Australia, including distribution and processes. The details obtained from the case studies and from other relevant literature have been used to assess the potential for future major impacts from nitrate contamination of groundwater and to address the future directions needed to better characterise the problem and its significance.

Nitrate contamination data

The understanding of the extent of nitrate contamination of groundwater and the processes causing it has been developed in Australia largely from a randomly collected series of data. Some targeted studies have been undertaken over limited time frames or in relatively small areas, and provide a snapshot of information or a localised evaluation of nitrate contamination.

There is limited routine monitoring of nitrate in groundwater and there are numerous uncertainties regarding the nature of the data available. Time series data are typically inadequate to establish seasonal and other variability in groundwater nitrate contamination even within the same land use area. Where there have been repeat sampling events, there is often inconsistency in the results obtained even from the same bore. Many of the nitrate analyses available have been obtained from water samples taken at the initial construction of a bore.

There are also concerns regarding the quality of the early analyses and the manner in which results are reported. There is still no consistency in the reporting of nitrate concentration although most workers report concentrations of NO_3 as N rather than NO_3 as NO_3^- .

Nitrate contamination extent, impacts and trends

The study has found that contamination of groundwater by nitrate in Australia is widespread and is associated with a wide range of nitrogen sources. The sources occur in both rural and urban environments. Many of these sources have been managed in such a way that excessive loads of nitrate have migrated below the soil zone and reached underlying aquifers.

There are three categories of nitrate sources which vary in their spatial distribution and loads to the groundwater system. These are:

- Broad area sources, such as grazing, dairying and fertiliser applications which have the potential to affect large areas with widespread nitrate loads;
- Multiple point sources, such as animal husbandry, effluent disposal and septic tanks which form single point sources or broad scale sources when their effects are aggregated; and
- Naturally occurring nitrate sources such as termite mounds and nitrogen fixing native vegetation.

Background nitrate concentrations in groundwater across Australia are in the order of less than 2 mg/L NO₃ (as N). However, after the impacts of the above sources, in many areas the concentration is greater than the *Australian Drinking Water Guidelines* (NHMRC–ARMCANZ, 1996) recommended maximum concentration of 10 mg/L and makes the groundwater resource in these areas unfit for drinking. In some of the more contaminated areas nitrate concentrations exceed 100 mg/L.

The highest concentrations of nitrate occur more commonly in shallow unconfined aquifers which are most susceptible to contamination. However, there are numerous instances of nitrate impact at depths of 50 m or more.

The extent of mixing and deeper impact depends on local aquifer properties, the groundwater flow system characteristics and time. Depth aspects need to be carefully considered in establishing multi-level bores for monitoring. Statistical analysis and numerical modelling is likely to assist in designing monitoring networks.

As well as impacting on direct use of groundwater, high nitrate concentrations can affect the water quality in receiving environments leading to eutrophication and development of algal blooms. This is an emerging problem particularly in areas of broad scale nitrate accession. The impact of nitrate on surface water systems is highly dependent on the relative contributions of surface and groundwater to the water body.

Nitrate contamination processes

The 10 case studies conducted have indicated that there is large variability in the conditions which yield high nitrate concentrations in groundwater. It appears that where a source of nitrogen exists there is the potential for nitrate to reach the groundwater beneath the source. The concentration of leachate which reaches the groundwater depends on local conditions at the source. The most appropriate means of minimising nitrate loads to groundwater is to carefully manage the application rates taking the local site factors into account.

The key factors which influence the ultimate nitrate load to the groundwater are:

- whether nitrogen can be used effectively at the surface by plants in the soil zone or there can be minimal leakage below the soil zone (particularly in the case of a point source);
- variations in soil type and recharge rate which control the rate of leachate migration through the soil zone;
- conditions in the soil zone and the unsaturated zone which can prevent the production of nitrate or denitrify nitrate, and cause adsorption to soil; and
- reduction of the nitrate concentration by the conditions in the aquifer which can allow denitrification or attenuation by other means such as dilution.

While there is a general understanding of the nitrate contamination processes in soil and groundwater, for any specific study or land management situation the characteristics of both the groundwater system and the unsaturated zone need to be carefully understood to evaluate nitrate contamination and its future impacts.

Potential for ongoing contamination

The scattered approach to data collection and the wide variability in aquifer response to nitrate loadings has not clearly identified trends in nitrate contamination in groundwater. However, it is commonly accepted that nitrate generally behaves as a conservative solute. Once it enters the soil and groundwater system there is very limited potential for a reduction in the nitrate concentration. Therefore, if current management of nitrogen sources continues, there is potential for increased accession of nitrate to aquifers and increases in both nitrate concentrations and in the extent of aquifers affected.

Major considerations in evaluating the potential for future impacts are:

- Significant loads have been applied over broad areas from grazing, cultivation and fertiliser application. With a continuation of current farming practices in Australia there is unlikely to be a significant reduction in environmental nitrate loads available to migrate to the watertable. Changes to management practices and education will be required;
- The ongoing clearing of land for both urban and rural development represents a potentially increasing source of nitrates from a variety of both diffuse and point sources; and
- Multiple point sources which apply significant loads have led to high nitrate concentrations in groundwater. Improved management of such nitrogen sources could reduce the ongoing release of nitrate into the groundwater system. Developments such as manufacturing and processing of agricultural products, solid waste and effluent disposal typically require regulatory approval. On site operational restrictions have the potential to reduce the actual load of nitrate for disposal.

Recommendations

This study has identified a broad range of issues associated with the existing and future management of nitrate in groundwater in Australia. The following recommendations are put forward to establish management actions, and to reduce gaps in our existing knowledge.

Management and policy development

- Change the focus of policy and research on nitrate contamination in groundwater from point sources of nutrients (which can be managed) to broad area diffuse sources (which require more complex management).
- Develop guidelines for groundwater protection zones around major potable water supply areas specifically focusing on nitrate sources.
- Coordinate with State Agencies to develop programs for land management which improve nutrient applications in broad area farming.
- Encourage the COAG Water Reform Committee to more actively include water quality (particularly nitrate) in developing future policy decisions.

Confirmation of nitrate trends

- Develop suitable groundwater monitoring networks in key areas where nitrate contamination from existing land use is already known. This will need to be conducted in association with State Agencies. Typical priority areas based on the Nitrate Map prepared for this study include:
 - Toowoomba;
 - Perth coastal plain; and
 - broad area grazing country in Victoria, NSW and SE South Australia.

- Emphasis to be placed on fully categorising the extent of nitrate pollution laterally and vertically.
- Evaluate any long-term urban water supply monitoring data on nitrate concentrations to establish nitrate trends. This will incorporate information in many areas around the country which are outside the study areas in this project.

Research activities

- Establish techniques for identifying risk to groundwater from nitrate pollution around key industries which can produce high nutrient loads.

Two major tasks are proposed:

1. Risk mapping in areas of broad area nutrient loading; and
2. Development of a 'pollution index' for more localised nitrate source management, eg. point sources, taking into consideration factors such as watertable depth, lithology, groundwater flow system and groundwater beneficial use.

Priority areas for these activities are:

- areas of dairying;
- areas of high application of fertilisers for horticulture over broad areas;
- areas where there are urban water supplies; and
- high value groundwater sourced waterbodies and ecosystems.
- Establish research programs into denitrification, including determine the key processes and conditions allowing denitrification. The relationship of denitrification to climate is an important process.
- Establish trial sites in areas where land management is changing and monitor impacts on nutrients in both soils and groundwater, eg. areas in SW Victoria which are soon to be or have recently been changed from forested to agricultural land use.

Funding allocations

As a guide to LWRRDC on the way in which it should meet these recommendations, the allocation of responsibilities for funding and the priority for each task is presented in Table 1.

Table 1
Responsibilities and priorities for future works

Item	Priority for LWRRDC	LWRRDC %	State Agencies %
Monitoring	5	5	95
Research			
Risk Mapping	2	30	70
Denitrification Studies	3	100	
Establishment of Trial Sites	4	40	60
Management and Policy			
Develop Protection Guidelines	1	50	50

Introduction

Study organisation

This study has been undertaken on behalf the Land and Water Resources Research and Development Corporation (LWRRDC) to evaluate the significance of nitrate contamination in groundwater in Australia. The project has received additional financial support from several State agencies.

The project team comprised:

- Paul Bolger, Melita Stevens, Geoff Mason and Richard Evans (Sinclair Knight Merz);
- Peter Dillon and Claus Otto (CSIRO Centre for Groundwater Studies); and
- Mike Williams and Jaswant Jiwan (Department of Land and Water Conservation NSW).

Additional support has been received in the provision of data and discussion from the following individuals and organisations:

- Andrew Shugg (Natural Resources and Environment Victoria);
- John Hillier and David Free (Queensland Department of Natural Resources);
- Peter Jolly and Steven Tickell (Power and Water Authority Northern Territory); and
- Brian Keating, Keith Weier and Kirsten Verburg (CSIRO Tropical Agriculture).

Study aims

The overall objective of this study was to provide an overview of the current conditions of groundwater systems in Australia with respect to nitrate contamination and to evaluate the potential for ongoing problems of nitrate contamination.

The study aimed to produce the following outputs:

- Assessment of the significance of nitrate contamination in groundwater, and the level of risk posed by the combination of land use and groundwater conditions;
- Recommendations for further work on groundwater nitrate contamination processes, including generation of data, research and establishment of groundwater monitoring programs; and
- Ranking of key issues for future research and investigation of nitrate contamination and nitrogen source management for dissemination to community and resource managers.

This report presents an evaluation of what is known about the current status of nitrate contamination in Australia, including distribution and processes, what future directions need to be taken to better characterise the problem and its significance, and an assessment of the potential for future major impacts.

No fundamental research has been undertaken for the project, although there has been detailed evaluation of the literature and available regional data.

Nitrate contamination—background

Nitrate contamination in groundwater—worldwide

Nitrate has been reported above background concentrations in groundwater worldwide and it has been identified to be the most common and widespread chemical contaminant in groundwater (Spalding and Exner, 1993). Background levels of nitrate (as N) in natural groundwaters are typically low. Concentrations between 0.45 and 2.0 mg/L have been reported in groundwaters in Europe and the USA (Hallberg, 1989; Juergens-Gschwind, 1989) and from 1.15 to 2.3 mg/L in Australia (Lawrence, 1983).

Spalding and Exner (1993) refer to the work of Madison and Burnett, who in 1985, compiled a map of nitrate concentrations in groundwater from analytical data collected over 25 years from more than 87,000 wells. This represented the first comprehensive evaluation of the extent of nitrate contamination of groundwater in the USA. Madison and Burnett reported that the background level of nitrate in aquifers was greater than 3 mg/L ($\text{NO}_3\text{-N}$) in 15 USA states.

In Europe there is significant contamination of drinking water supplies which has been noted at levels of concern since the 1980s. In the UK this has resulted in the development of thirty *Nitrate Sensitive Areas* in which there is compensation provided for farmers who undertake improved management practices. In addition, groundwater management in the UK includes identification of groundwater protection zones around well heads to minimise groundwater contamination by nitrate and other contaminants, particularly those associated with agricultural practices.

Nitrate is also a common problem in island settings where there is little wellhead protection of the groundwater resource (Abell, 1993; Dillon, 1997).

In Australia, there is extensive occurrence of nitrate above the background levels. This report describes the nature and extent of the nitrate contamination in Australia, including a map of high (>10 mg/L as N) nitrate areas.

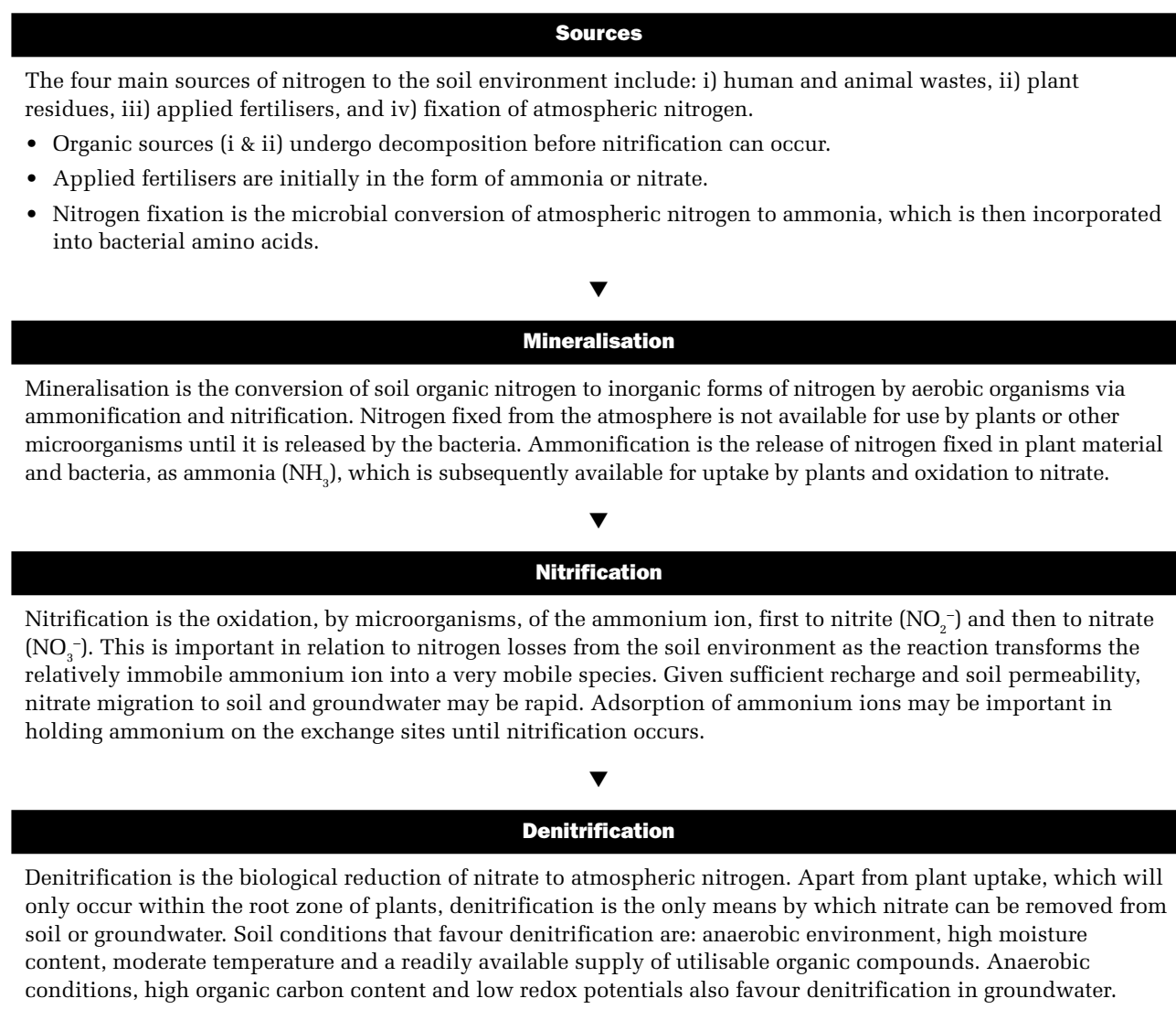
Introduction of nitrate to aquifers

Nitrate behaviour

Nitrate enters the groundwater after it is oxidised from other forms of nitrogen. This process may occur naturally or as a result of human activity. Numerous factors affect both the conversion of nitrogen to nitrate and its consequent migration to the groundwater system.

Nitrogen is an ubiquitous element in our environment and is present in the air, soil and water environments. An analysis of nitrate in the environment requires an understanding of the behaviour of nitrogen. The various processes affecting the entry of nitrate to groundwater are briefly outlined in Figure 1.

Figure 1
Summary of key nitrate generation processes



Once nitrate has been generated and passes below the plant root zone, it typically behaves as a conservative contaminant. Denitrification may occur under anaerobic conditions in the presence of organic carbon. However such conditions are not common in the unsaturated zone above the watertable and therefore over time, any nitrate load below the root zone is likely to reach the watertable.

Nitrate sources

Nitrate can enter groundwater from either diffuse or point sources of nitrogen. Diffuse (or non-point) sources are generally those where the origin of the contamination is not able to be accurately traced to a single polluter, or where the contamination arises from a number of closely-spaced similar activities.

Point sources of pollution are those where the origin of contamination can be identified. Examples of point sources of nitrate pollution include direct injection of effluent into soils and aquifers, localised agricultural practices that affect aquifers directly below the site (eg. feedlots), septic tanks and landfills.

Contamination of groundwater by nitrate from diffuse sources is commonly detected in aquifers less than 30 m deep (Hallberg, 1989). This is because the major nitrate sources occur at the surface and there is a delay in the migration of nitrate from the source to the groundwater system. However there are situations where elevated nitrate from surface sources is detected at greater depths. This will depend on the degree of mixing of the contamination in the aquifer and the time over which the contamination has continued.

In cases of direct injection of nitrogenous waste into the aquifer, the presence of nitrate is clearly independent of depth and is affected by conditions in the aquifer.

The key activities which can generate or mobilise sources of nitrate in Australia and lead to contamination of groundwater are:

Natural sources

- rainfall;
- degradation of natural vegetation; and
- termites.

Human activities mobilising natural sources

- influence of clearing/stocking on natural environments;
- tillage of soils; and
- cultivation.

Human introduced sources

- human waste water treatment;
- landfills;
- septic tanks;
- fertilisers;
- nitrogen fixing pastures;
- farm/feedlot animal wastes;
- sullage and storm water;
- industrial waste; and
- general urban development.

Nitrate contamination in Australia

Beneficial use and acceptable nitrogen concentrations

The significance of nitrate (or any other constituents) in groundwater in Australia is identified by its impact on the beneficial use of the water.

The 1992 Draft *Guidelines for Groundwater Protection* (AWRC 1992), describe the following beneficial use classes to be protected:

- human consumption/food production;
- agricultural, industrial and mining;
- ecosystem support; and
- no definable beneficial use/controlled degradation.

These draft guidelines do not associate specific water quality criteria with classes of beneficial use. Groundwater protection classes can be found in the ANZECC (1992) *Australian Water Quality Guidelines for Fresh and Marine Waters*. The relevant environmental values to be considered for groundwater in these guidelines are:

- ecosystem protection;
- raw water for drinking water supply;
- agricultural water; and
- industrial water.

The protection of groundwater and surface water is based on the principle that the existing or potential beneficial use of the water should not be impaired by any activity (ANZECC, 1992). The beneficial use of the water is considered impaired if the total dissolved solids (TDS) or other constituent concentration is raised above what is specified for the given beneficial use.

The recommended acceptable concentrations of nitrate for various beneficial uses are shown in Table 2. There are no guideline levels of nitrate and nitrite nominated for water used for agricultural irrigation.

Distribution of elevated nitrate concentrations in Australia

For the purpose of this study, a concentration of 10 mg/L nitrate (as N) has been adopted as a level which indicates that groundwater has been contaminated by nitrate. It is recognised that there are certain situations where such concentrations occur naturally (Lawrence, 1983) although many are related to human activity.

This nitrate concentration in groundwater is adopted on the basis of direct potable use of groundwater. Different criteria apply to ecological impacts, especially during periods of low stream flows or lake levels when groundwater baseflows are likely to be greatest.

There have been several reviews and books published on the issue of Australian groundwater resources contaminated by nitrate (Lawrence, 1983). In 1983, over 1,200 analyses of groundwater impacted by nitrate were compiled to indicate the extent of groundwater contamination with nitrate (Lawrence, 1983). Groundwater with significant nitrate contamination was identified in all Australian States and Territories.

Table 2
Acceptable nitrogen concentrations for beneficial use

Beneficial Use	Nitrate	Nitrite	Ammonium	Total Nitrogen
	mg/L (as N)			
Potable Water	10	1.0	0.01	—
Stock Watering*				
Horses	30	10	—	—
Sheep	60	10	—	—
Cattle	40	10	—	—
Industrial				
Hydroelectric Power Generation	—	—	—	0.5
Food and Beverage				
Brewing	<10	—	—	—
Dairy	<20	—	—	—
Food Canning	<10	—	—	—

Note: A dash denotes no guideline values nominated.

* Stockwatering guideline is 30 mg/L. The suggested values for sheep, horses and cattle are comments in ANZECC (1992) report. The extent of high nitrate concentrations (>10 mg/L as N) in Australian aquifers is shown in Figure 2. Data sets to compile the map were made available by relevant State Agencies. Most of the data on nitrate concentrations are from shallow aquifers and typically have been obtained after 1960.

Source: ANZECC (1992)

For all data sets available (excluding South Australia) the map shows that nitrate concentrations above 10 mg/L (as N) are widespread. The only nitrate concentration data made available from South Australia for the study was from the south east region. The distribution of nitrate in this state has not been thoroughly identified in this study. However it can be concluded that there are elevated nitrate concentrations in the groundwater across the nation in areas of differing land use, including natural settings and those affected by human activities.

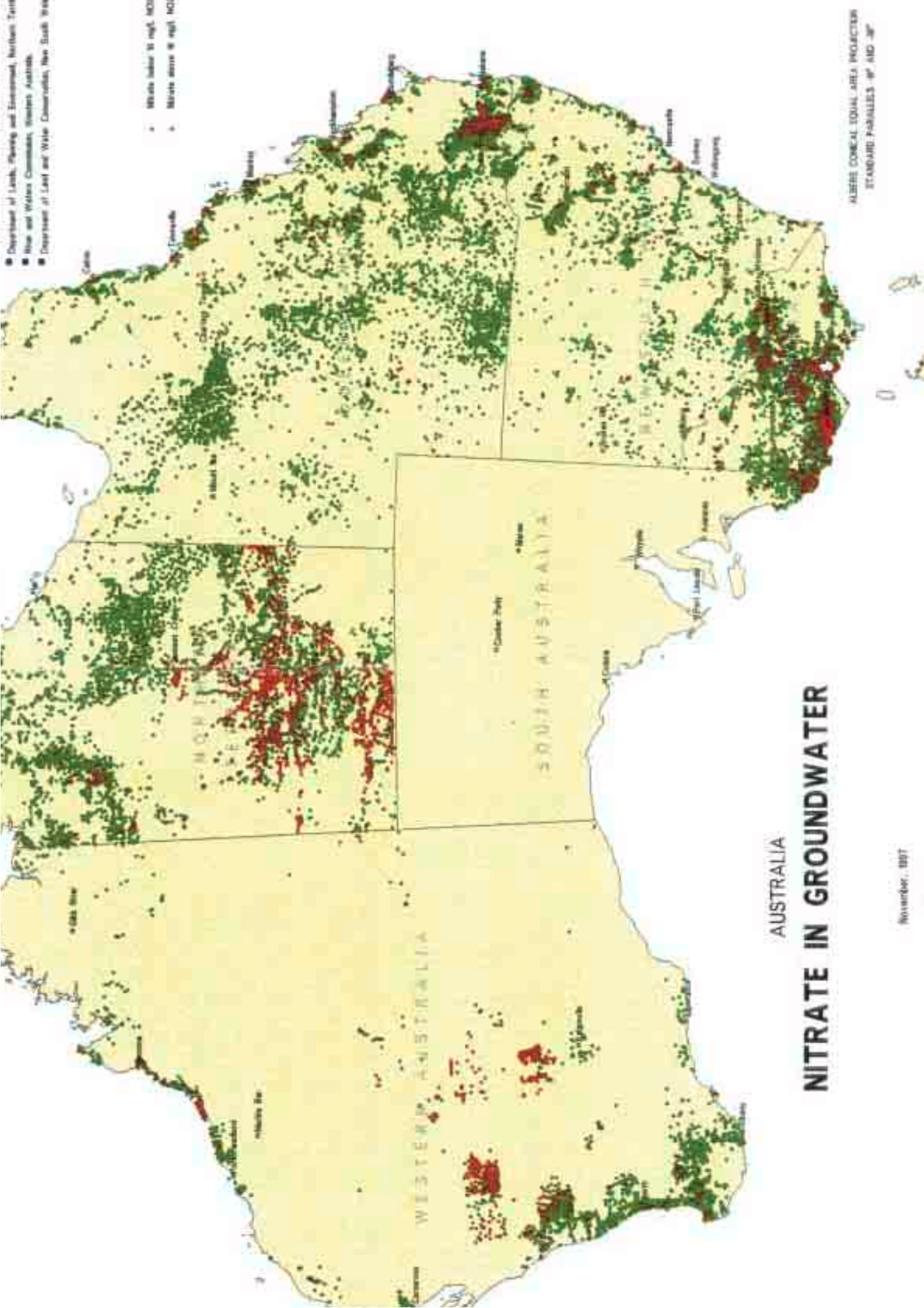
In a particular region, there are large numbers of bores from which groundwater samples have indicated the presence of aquifers in which nitrate concentrations exceed acceptable limits (>10 mg/L). For example, in 1991, one-third of all bores sampled on the Mount Gambier Plain, in South Australia contained nitrate levels in excess of 10 mg/L (Dillon *et al.*, 1991).

Problems arising from nitrate contamination

Direct use of nitrate contaminated groundwater

Elevated concentrations of nitrate in groundwater destined for potable or stock watering use represents a health risk to human or animal consumers.

The human health risk associated with consumption of water containing nitrate is due to the reduction of nitrate to nitrite in the human gut. Nitrite toxicity arises from the oxidation of normal haemoglobin to methaemoglobin which has impaired oxygen transport abilities, resulting in a condition called methaemoglobinaemia. Young infants, pregnant women and the elderly are more susceptible to methaemoglobinaemia than adults. Consumption of drinking water containing elevated concentrations of nitrate has been reported to cause the death of an infant in South Dakota, USA (Johnson *et al.*, 1987).



Department of Land, Planning and Environment, Northern Territory
New and Western Communities, Northern Australia
Department of Land and Water Conservation, New South Wales

Albers Conical Equal Area Projection
Standard Parallels 40° and 30°

November, 1997

Studies in laboratory animals have not indicated that nitrate or nitrite are directly carcinogenic, but there is evidence that they may react in the stomach with foods containing secondary amines to produce N-nitroso compounds which are known to be carcinogenic in animals (NHMRC-ARMCANZ, 1996).

There have been non-confirmed relationships reported between ingestion of nitrate in drinking water and a number of human health conditions including hypertension, increased infant mortality, central nervous system birth defects and stomach cancer (Spalding and Exner, 1993).

The guideline level recommended in the recently revised *Australian Drinking Water Guidelines* (NHMRC-ARMCANZ, 1996), of 50 mg/L nitrate (as nitrate) (11.3 mg/L NO₃ as N) has been derived to protect the health of young infants. It is stated that up to 100 mg/L (as nitrate) can be consumed by adults and children over three months of age without significant health effects.

Ecological impacts of nitrate contaminated groundwater

Nitrate in groundwater can represent a risk to the environment from nutrient addition where groundwater discharges into surface watercourses. The increased loading to surface water systems from discharge of nitrate contaminated groundwater can alter the nitrogen:phosphorus ratio in the receiving water, resulting in an increased risk of eutrophication and algal blooms.

There appears to be limited investigations on the ecological impacts of nitrate contamination of groundwater on surface waters, despite the preparation in recent times of various Nutrient Management Strategies (eg. DNRE, 1996). Surface and groundwater monitoring and interaction are often considered separately. The recent NSW Groundwater Policy recognises the interaction of the surface and groundwaters and it could be expected that nitrate contamination could be considered as part of the overall groundwater protection and management.

There is not a large amount of data on the impact of nitrate on ecological sites, although extensive algal blooms which occurred in the Darling and other rivers in Australia in 1991 (Verhoeven, 1992) indicate the presence of high concentrations of nutrients as well as other parameters. There is a general lack of data on eutrophication in Australian waters, and as a result there are no maximum nitrogen levels set for control of eutrophication.

In terms of acceptable nutrient levels for Ecosystems, the *Australian Water Quality Guidelines* (ANZECC, 1992) consider that site specific studies determine the nutrient loads to streams. This should be based on the stream baseflow (ie. the groundwater contribution of streamflow). However indicative figures for rivers and streams in Victoria are total nitrogen of 0.1–0.75 mg/L. Where nitrogen levels exceed 1 mg/L in Victorian waters, the riverine environment is considered to be degraded (Tiller and Newell, 1995).

With the very wide distribution of elevated nitrate in groundwater from many sources throughout Australia, the impact of discharge of nitrate contaminated groundwater and the siting of potentially contaminating activities close to receiving waters is of great significance.

Case studies

Approach to case studies

10 case studies have been used to develop an appreciation of the key nitrate contamination processes and draw conclusions on the significance of the key issues for nitrate contamination of groundwater in Australia. The case study results form the basis for recommendations for future investigation and management.

The following sections outline:

- The basis for selection of the study areas;
- Summaries of the case study results (detailed descriptions of each of the study areas are presented on pages 54–134); and
- Conclusions drawn from the case studies.

Selection of study areas

The various case study areas were selected to investigate a variety of different situations which potentially result in groundwater contamination by nitrate. The approach taken has been to identify the most suitable study area to characterise the key issues for a type of nitrate source.

In addition to the specific case studies, information was also obtained from other regions with comparable sources of nitrate contamination. These supplementary sources of information were used to add to the understanding of the processes and the significance of the contamination for the various land uses or aquifer conditions.

The study areas and a brief description of the land use, nitrate source and soil type are listed in Table 3.

The following criteria were used to select the major study areas:

- Relative significance of nitrate contamination;
- Land use(s) in the study area;
- Hydrogeological setting;
- Abundance of data; and
- Geographic location and climatic zone.

Relative significance of the nitrate problem

Initial screening of the study areas involved identifying areas where:

- the groundwater nitrate concentration was known to be or thought to be likely to exceed 10 mg/L (as N); and
- there was high demand for groundwater or the potential for major impacts on surface water systems through discharge of nitrate contaminated groundwater.

Land use

Land use takes into account the nature of the nitrate source. It includes the typical loading of nitrate generated from the land use, and the distribution of the nitrate as either a point source or distributed source. The following land uses were considered in this study:

- Grazing (Case Studies 1, 9);

- Cropping (Case Studies 9, 10);
- Effluent disposal (Case Study 2);
- Septic tanks (Case Study 3);
- Intensive horticulture, irrigated and dryland (Case Studies 5, 8, 9, 10);
- Natural Processes, eg. termite mounds in central Australia (Case Study 4);
- Urban activity (Case Studies 6, 7, 10); and
- Animal husbandry (Case Studies 5, 9).

In several areas there is mixed land use associated with groundwater nitrate problems.

Hydrogeological setting

High nitrate concentrations have been observed in groundwater in a wide range of aquifer types. This project attempts to identify case study areas which include as many of the key aquifer types known to have been affected. This has included both porous media and fractured rock systems. The latter are commonly basaltic terrains.

The typical lithologies for each case study area are shown in Table 3.

Abundance of investigation data

The case studies selected have included areas where there is a substantial amount of monitoring data or for which there has previously been a substantial amount of detailed investigation.

Geographic location and climatic zone

The study areas have been selected to cover a wide a range of geographic and climatic conditions.

While most information is available for temperate climates, significant nitrate concentrations occur in tropical (Case Study 8) and arid (Case Study 4) environments.

Table 3
Study areas selected for detailed investigation

Area	Lithology	Land Use (ie. Nitrate Source)
1. SE South Australia (Shepparton (Tongala))	sands/limestone clays	grazing grazing on irrigated pasture
2. Werribee	clays	effluent disposal
3. Venus Bay, Nepean Peninsula (Benalla) (Toowoomba)	sands alluvial deposits basalts	septic tanks
4. Ti Tree Basin	sands, clays	‘natural arid zone processes’
5. Jandakot	sands	Mixed agricultural includes intensive horticulture, grazing, animal husbandry and poultry farming (denitrification a key issue)
6. Perth Metropolitan (Warrnambool)	sand limestone, basalts	urban
7. Wagga	alluvium, siltstone, slate, phyllite	urban and rural residential
8. Bundaberg (Shepparton East)	sand sands	tropical intensive horticulture (sugar cane) irrigated intensive horticulture
9. Narromine	alluvium	irrigated cropping (grazing urban)
10. Peel Valley (Toowoomba)	shales, sandstone, fractured granite basalt	grazing, dryland and irrigated cropping and intensive animal husbandry intensive animal husbandry

Areas and conditions in brackets are considered to be ‘supplementary’ areas have been identified which provide further information on the behaviour of nitrate in groundwater systems.

Case studies

Each of the case studies undertaken for this investigation is summarised in the following tables to provide an overview of the main issues which are relevant to the understanding of the status and impacts of nitrate in Australia.

Detailed discussion of the major study areas and some of the supplementary areas including relevant maps, bore plots, and data sources and references are presented on pages 54–134 of this report.

The conclusions drawn from the case studies are outlined in the following two chapters. They include an assessment of:

- the suitability of present monitoring of nitrate concentrations in groundwater;
- the significance of nitrate contamination in groundwater in Australian aquifers;
- the factors affecting existing and future nitrate contamination of groundwater; and
- the ranking of the most significant land use and site conditions affecting the concentration of nitrate in groundwater.

Case Study 1: South-East South Australia— Pasture, mixed agriculture and forestry

This case study is from a pastoral and agricultural area centred around Mt Gambier in South Australia. Agriculture has occurred since the mid 1800s. The case study has considered the agricultural impacts rather than the effects of the multiple point sources which are widespread, particularly in the vicinity of the urban area of Mt Gambier.

The major aquifer in the area is the Tertiary Gambier Limestone. This outcrops extensively and is overlain by aeolian sands. The soils developed are generally shallow and permeable. Recharge rates to the shallow aquifer are relatively high.

The major diffuse nitrate sources in this study area result from dryland pastures, field cropping, and irrigated cropping. Additional land uses include pine plantations, market gardening, viticulture, and potatoes. Nitrate concentrations in the groundwater in the Gambier Limestone are frequently above 10 mg/L NO₃ as N. There is localised evidence of denitrification, particularly in shallow zones of organic rich sediments. Elsewhere in the Gambier Limestone the aquifer is aerobic, the nitrate behaves conservatively and is able to migrate through the aquifer with the groundwater flow. In contrast, any leakage of nitrate from the Gambier Limestone to the deeper aquifer (the Dilwyn Formation) is not likely to be conserved because of the anaerobic conditions in this aquifer.

This study area has indicated that:

- The nitrate concentrations above 10 mg/L occur under grazing on irrigated and dryland dairy pastures. A significant source of the nitrate is urea from cattle urinating in paddocks.
- Leaching of nitrate occurs at high rates in areas of soil tillage where pasture improvement or cropping occurs. This may be increased by irrigation.
- The highest concentrations of nitrate are in the upper intervals of the aquifer (mainly less than 50 m).
- Significant nitrate concentrations occur at the many point sources around the area, particularly in the vicinity of Mt Gambier. The high point source loads introduced over time progressively provide additional mass of nitrate to the aquifer. Likewise reduction in loads for older sources and the use of stormwater drainage bores distorts the nitrate monitoring trend in and around the urban area.
- Limited monitoring data over the broader area of the south-east in the main areas of grazing does not show a strong correlation of the nitrate concentrations with time. Monitoring over ten years (1972–82) has shown no apparent increase in nitrate.
- Analytical modelling suggests that with the increasing mass of N into the aquifer and limited opportunity for denitrification there could be increases in nitrate concentrations well above guideline levels for drinking water over large areas. The extent of mixing in the aquifer and the ultimate vertical distribution of nitrate is not known.

Key reference: Schmidt *et al.*, 1998

Case Study 2: Werribee, Victoria— Irrigation with municipal effluent

This case study involved the flood irrigation of pasture paddocks with raw municipal effluent and an investigation of the mechanisms which result in the generation of leachate below the root zone and migration of nitrogen compounds to the underlying watertable.

The study site comprises a clay soil overlying basalt. The site is situated on a coastal aquifer which is confined to semi-confined with a thickness that varies up to 40 m. At the trial site, the watertable occurs at a depth of approximately 1.5 m. Groundwater quality is of potable to irrigation quality and used extensively for market garden irrigation.

The nitrate sources at the study site arise primarily from municipal effluent disposal with average raw effluent nitrogen concentration of 41 mg/L, with 31 mg/L of the Total Nitrogen (TN) being in the form of ammonium. Over the trial plot, the nitrogen load per irrigation event is 2.56 kg/ha, which equates to 256 kg/ha/yr. A second source of nitrate is from animal manure and urine from cattle and sheep which are allowed onto the site to graze between irrigation events.

Radio-labelled ammonium sulfate was mixed with raw effluent and added to mini-lysimeters over three irrigation events. Drainage water was collected from lysimeters which were destructively sampled at the completion of the experiment to determine fate of nitrogen.

Study results showed that:

- 2% of the applied labelled N leached, with concentrations of up to 45 mg/L in leachate;
- 35% of labelled nitrogen was retained by the soil, 90% of which was incorporated into soil organic N;
- 19% of labelled nitrogen was recovered from plants; and
- 44% of labelled nitrogen was unaccounted for, assumed to be lost by denitrification or gaseous loss.

The leaching of nitrate in clay soils with relatively high watertable, plus probable denitrification accounts for only small percentage of nitrogen leaching and a large unaccounted for nitrogen fraction. Regardless of this low percentage of leachate, the nitrate concentration in underlying groundwaters at 10 m depths is elevated to more than 10 mg/L. Elsewhere in the area of long-term effluent disposal at Werribee, there is little nitrate and ammonia in the dominant form of nitrogen.

Additional information on irrigation of treated municipal effluent is obtained from the Wagga Wagga Effluent Treatment plant. This covers disposal of treated municipal effluent in a large-scale program. Irrigation of trees is carried out and the trees are utilising the nutrients and the water in the effluent. There is an excess of nitrate above plant water requirements, leading to generation of elevated groundwater nitrate concentrations (>10 mg/L as N).

Key reference: DNRE, 1997

Case Study 3: Venus Bay and Nepean Peninsula, Victoria— Septic tank

Both the Venus Bay and Nepean Peninsula study areas comprise coastal sandy dune deposits with large numbers of septic tanks installed in holiday houses. The septic tank densities in specific are greater than 15/km² in some areas, and this has resulted in nitrate concentrations of >10 mg/L at numerous locations. Septic tanks have been installed in these unsewered areas for many years.

The groundwater is typically of potable quality and used for domestic purposes. This reflects the direct infiltration recharge which occurs in these deposits. It is also an indication of the high potential for rapid recharge of septic tank nitrate in the aquifer. Groundwater conditions are typically aerobic with little opportunity for denitrification.

The data for these areas enable the pattern of aerial distribution to be mapped, although there is a poor temporal distribution of data. Therefore trends cannot be determined. However the rapid rate of infiltration suggests that while there is a source of nitrate at a septic tank, there is little likelihood of a reduction in nitrate concentration. Perhaps some reduction in concentration during high recharge events might occur as a result of dilution.

Additional information on septic tanks as a source of nitrate pollution to groundwater is indicated by data from Benalla in northern Victoria where there is a thin alluvial gravel aquifer overlain by 5 m clay deposits. The nitrate concentration in this area reaches up to 16 mg/L (as N), where the septic tank density is greater than 15/km². However this is based on only one set of data.

There is evidence that the abundance of septic tanks in basalt aquifers in the Toowoomba area (Queensland) has led to nitrate concentrations well above 10 mg/L. This suggests rapid infiltration of nitrate through the fractured rock medium in to the watertable.

Key reference: SE Water, 1996; HydroTechnology, 1993.

Case Study 4: Yulara, Northern Territory— Natural process nitrate contamination

Yulara is a study area in Australia's arid zone which is underlain by Proterozoic to Cambrian rocks overlain by Tertiary sediments up to 100 m thick, comprising sand, clay, some calcrete and lignite. The major aquifers are sand layers in the Tertiary sequence and are unconfined or partly confined. The depth to watertable decreases from 30 m south of the study site to 15 m.

Nitrate sources were considered to be non-anthropogenic due to low population density and lack of agricultural activity. Groundwater quality is generally less than 1,500 mg/L TDS, but ranges from 1,500–5,000 mg/L TDS. 30 observation bores were sampled and nitrate concentration exceeded 10 mg/L (as N) in most bores. The maximum nitrate concentration recorded was 54 mg/L.

Examination of the distribution of high nitrate concentrations at or near the soil surface showed a number of possible nitrate sources—the highest soil concentrations of nitrate were found in termite mounds, with surface soil concentrations of up to 2,000 mg/L measured. Nitrate was also found in spinifex and other grasses, leaf litter in mulga, in surface crusts and in bare sandy soil covered with bushfire ash. The presence of organic matter (especially in termite mounds) was considered the major factor for nitrate production

The study concluded that:

- bacteria within termite mounds fix nitrogen to ammonium which is oxidised by other bacteria to nitrate and leached out of the mounds;
- cyanobacteria in dry surface soil produce nitrate after rainfall events;
- recharge pulses from heavy one in 20 year rainfall result in flushing large quantities of nitrate to groundwater;
- the arid conditions at the site result in lack of denitrifying bacteria in the soil; and
- fire plays an important role in increasing the available nitrogen at the soil surface.

Key reference: Barnes *et al.*, 1992

Case Study 5: Jandakot Mound, Perth, WA— Mixed agricultural

This case study is based on the mixed agricultural and rural residential area of the Jandakot Groundwater mound, south of Perth, WA. The main aquifer comprises the surficial sand aquifers of the Perth Coastal Plain. The groundwater mound is an important recharge area for groundwater sourced water supply to the City of Perth. These deposits are therefore in need of particular protection of the water quality.

The land uses in the vicinity of the Jandakot Mound are residential, rural smallholdings (eg. kennel zones), horticulture, floriculture, intensive animal industries, industrial and commercial activities, parks and ovals, and remnant bushland. The Jandakot mound is located within the Peel–Harvey coastal catchment and therefore land uses here impact not only on groundwater quality, but also on eutrophication of the Peel–Harvey Estuary. Various planning decisions have been made to protect this valuable resource from degradation in quality and quantity.

Nitrate concentrations in the groundwater in the area are generally low (<1 mg/L NO₃ as N.) although there are concentrations as high as 100 mg/L in certain areas. There appears to be elevated concentrations of nitrate associated with septic tank and other disposal of waste from Kennels which are widespread in the area. Groundwater sampled from public and private bores in the area suggests that the high concentrations are local and that there is not significant migration of nitrate to nearby Water Corporation production wells even though there is potential for this to occur. Recent work suggests that there is likely to be denitrification in the groundwater at the Jandakot Mound.

Trends of groundwater quality data for 204 bores for the last ten years do not show a consistent pattern of significant increases in the groundwater nitrate concentration. However the records do show very sharp changes in nitrate concentration within a short period of time.

Key reference: EPA WA, 1991; EPA WA, 1993

Case Study 6: Perth Metropolitan— Urban land use

Groundwater drawn from unconfined shallow aquifers on the Swan Coastal Plain comprise 75% of the water used in the Perth Metropolitan area. There are also numerous groundwater fed wetlands across the Swan Coastal plain. The groundwater beneficial use is both for direct use and ecosystem protection.

Soils in the region are typically sandy and have poor soil water and nutrient retention capacity. The shallow aquifers are therefore susceptible to contamination from surface and near surface processes.

Urban development in the Perth Metropolitan area has taken place rapidly with significant land use changes, including alteration of natural bushland. As it is anticipated that groundwater resource development will increase in the future, zoning of groundwater protection areas has been implemented to maintain groundwater quality.

Nitrate concentrations in surficial aquifers in the Perth Metropolitan area are elevated and there is evidence that the expansion in urbanisation is causing the concentrations to increase. However there is also evidence that the presence of organic carbon, as well as other reducing agents such as sulphides and Fe(II) minerals, is causing denitrification. Local variations in the nature of surficial deposits results in different nutrient retention and denitrification potential. Therefore in some parts of the Perth Metropolitan area (eg. in areas of the Bassendean Sands) the nitrate concentrations are not as high as may be expected.

Variations in the physical and chemical characteristics of the groundwater systems have been shown to affect the nitrate concentrations in the groundwater. Increasing nitrate concentrations have been detected at Gwelup. Comparison of the higher nitrate groundwater system found at Gwelup where there is an open system with recharge to groundwaters high in dissolved oxygen, contrasts with low concentrations in the closed system at Mirrabooka where there is excess organic carbon, low redox, potential and anaerobic conditions which would favour denitrification. An understanding of the groundwater system is needed to understand the impact of land use changes and the introduction of nitrogen to the groundwater system.

There a number of major sources of nitrate in the Perth Metropolitan area.

Septic tanks are very common and comprise a concrete anaerobic digestion tank from which the treated effluent is discharged into soak wells or leach drains. In general, once the effluent passes through to the aerobic soil (usually within 0.5 m), it is oxidised to nitrate. Depending on the local site conditions, with the aerobic conditions and the low cation exchange capacity of the soil, nearly all of the nitrate can enter the groundwater. As would be expected, nitrogen loading rates in unsewered areas are higher than in sewerred areas and therefore greater potential for nitrate contamination exists in the former.

Horticulture/market gardening in the Perth region is a significant contributor to nitrogen loads from the application of fertilisers and the irrigation with nitrate contaminated groundwater. Many of the bores associated with horticultural activities have nitrate concentrations above 10 mg/L and as high as 50 mg/L NO_3 as N, although the elevated concentrations appear to be in the upper parts of the aquifer. As in the case of the septic tanks, the concentration depends on the denitrification capability of the local conditions. The leachate recovered from beneath the root zone of horticultural activities has been found to be comparable to the leachate from septic tanks. Lateral migration in the aquifers shows some variation with high concentrations observed downgradient in several horticultural areas, while there appears to be a relatively rapid decrease downgradient at other localities, again likely to be related to denitrification processes within the aquifer.

Gardens and lawns where fertilisers have been applied, both in private homes and in council parks, provide high nitrogen loads and have resulted in high nitrate concentrations in the aquifers in some areas. Concentrations as high as 20 mg/L NO₃ as N have been reported. With higher irrigation rates there is a higher rate of leaching and higher concentration of the leachate beneath the soil root zone has been observed for the same nitrogen application. The high concentrations in public and private gardens and lawns contrasts with the nitrate concentrations of around 1 mg/L NO₃ as N in forested, rural and native bush areas.

The general conclusion on future trends in groundwater contamination by nitrate from the Perth metropolitan study is that as development in urban areas expands in these sandy environments, there will be greater potential for nitrate to be introduced. As more areas become sewerred there will be a potential to reduce this source. However there will be mobilisation of any soil nitrogen by clearing and tillage of soils, particularly in the *Acacia* country along the coastal limestone belt, and increased supply from fertiliser application in urban gardens and lawns. In older areas there will need to be consideration of the integrity of the sewerage system and the potential for leakage to the unsaturated zone.

It is concluded that under conditions in which a continuing nitrate load is applied from the multitude of relatively unregulated sources, and there is little or no potential for denitrification, nitrate concentrations are likely to increase. Similarly with an expansion of clearing of land with the potential to provide a nitrate source, the extent of elevated nitrate concentrations will increase. The concentrations could be in order of those found around Gwelup.

A key finding from the Perth area is the impact on nitrate leachate of varying conditions in the unsaturated and saturated zones. In particular the distribution of organic matter, the redox potential and the possibility of denitrification needs to be understood. The impact of the groundwater flow system on the distribution of denitrifying zones needs to be understood. This indicates that for particular developments to be established, there is a need to establish the local conditions which influence what the aquifer can accept without detriment to quality.

Key reference: Barber *et al.*, 1991; Gerritse *et al.*, 1990; Sharma *et al.*, 1991, 1992, 1993; Appleyard, 1995

Case Study 7: Wagga Wagga— Regional urban

Wagga Wagga is a city of more than 56,000 population and is located in the Murrumbidgee Valley.

The city is underlain by slate and granite. There is an alluvial sequence up to 150 m thick associated with the Murrumbidgee River. It comprises a highly permeable sandstone gravel sequence which underlies a shallow clay band. Individual units may be up to 30 m thick.

The land use in the area varies. Wagga Wagga itself is an urban area. Dry cropping and grazing occurs in the surrounding area. An effluent treatment plant is located near the edge of the city and there is a landfill nearby.

There is also a feedlot within about 25 km of Wagga Wagga.

Six monthly monitoring of nitrate from monitoring of Town Water Supply Bores/extraction points operated by Southern Riverina County Council, for Wagga Wagga and surrounding areas indicate that nitrate concentrations in the region are low, the highest being 2.3 mg/L NO₃ as N. The water supply bores in the city of township of Wagga Wagga and the surrounding communities generally record NO₃ (as N) concentrations at less than 0.5 mg/L with occasional higher values up to 1.9 mg/L. In areas of dry cropping in several areas nitrate has been detected at concentrations up to 2.3 mg/L.

There is no strong indication of elevated nitrate concentrations for the city water supply or the areas around Wagga Wagga. This contrasts with the elevated nitrate noted in other regional urban areas. Dry cropping in the regions around Wagga Wagga show the highest concentrations of nitrate, even though they are low (<2.5 mg/L).

Case Study 8: Bundaberg, Queensland— Tropical intensive horticulture (sugar cane)

Intensive horticulture under tropical conditions using sugar cane as the major crop is practised in a number of areas along the Queensland coast. Significant nitrate contamination of groundwaters which is well documented in the Bundaberg area from a number of local and subregional monitoring studies, has been detected in a number of these groundwater systems.

The aquifers in the area comprise two Tertiary Sand units partially separated by a clay aquitard. Surficial sand deposits occur in coastal areas. Soil types in the area are typically sandy but vary in infiltration rate and resulting nitrate concentration.

Land use in the Bundaberg district has been dominated by sugar cane production for over 50 years. A small amount of other horticulture, often tomatoes, as well as some grazing also take place in the area. There is an expansion of urbanisation and rural residential development in the area with a resulting change in potential sources of nitrogen for groundwater contamination. The results of investigations show the presence of nitrate across all land uses. However there appears to be a substantial source of non-point nitrate contamination resulting from irrigation and fertiliser application to sugar cane in the Bundaberg area.

The mean concentration of nitrate (as NO_3) in groundwater from parts of the aquifer supplying the urban water supply is generally less than the health risk levels. However there are a number of rural areas where the elevated nitrate concentrations make the groundwater unsuitable for domestic use. ie. nitrate concentrations above acceptance criteria 45 mg/L NO_3 as NO_3 .

The main centres of high nitrate concentration coincide with the most loamy sols in the area and are in the highest recharge sites. The recharge mechanisms at Bundaberg appear to have a significant influence on the concentration of nitrate in the groundwater. This is reflected in the monitoring information which suggests that in some areas, there is a seasonal fluctuation in nitrate concentration, with lower concentrations recorded during periods of higher recharge.

The depth of bores containing high nitrate concentrations is variable with most wells intersecting the shallow Elliott Formation. There are however a number of high nitrate concentrations reported from bores at depths of greater than 50 m.

Long-term monitoring bores have a variable trends in nitrate concentration and there is no evidence of a regional increase in nitrate concentration with time. In fact the various monitoring bores show rising, falling and stable trends. However data from separate bores but within the same area suggest that there may have been an increase in nitrate concentration of 20–30% over 20 years. This trend would need to be confirmed by regular monitoring and integrated with details of the groundwater flow patterns and chemical characteristics.

The key source of nitrate appears to be the high applications of fertiliser, with increased accessions in areas of higher recharge. In addition there is some evidence that septic tanks provide a source in rural residential and in small urban areas.

A number of different climatic and hydrogeological zones where there is intensive horticulture for other crops provide a different perspective on the nitrate processes but there are fewer data. These are Shepparton East (VIC) where and Rochedale (QLD) where there is a range of cash crops.

Key reference: Keating *et al.*, 1996a; 1996b, 1997

Case Study 9: Peel Valley, NSW— Mixed agricultural land use

The study area is situated around the rural township of Tamworth in northern NSW. The geology of the surrounding area generally comprises meta-sediments covered with a thin alluvial layer, and granite with a thick residual regolith in some areas. Alluvial flats occur along the Peel River.

The watertable is at a depth of around 2–3 m in the alluvium, but appears to vary from around 5–12 m in the fractured metasediments and around 5 m in the granite.

The main source of groundwater in Peel Valley is unconsolidated alluvium of the Peel River and its tributaries. The alluvial system comprises shallow (less than 15 m) clays, silts, sands and gravels. Individual bore yields in the alluvium are less than 15 L/s.

There is a range of land uses in the Peel Valley. The predominant land uses are grazing, dry cropping, with other important industries irrigated cropping, intensive animal husbandry (mostly piggeries, poultry farms) and rural residential.

The overall pattern of nitrate contamination suggests that there are high concentrations of nitrate (>10 mg/L and including very high values of 52–170 mg/L NO₃ as N) associated with grazing in the area, particularly in the granite country where there are relatively permeable residual soils. In areas of lower permeability fractured metasedimentary rock aquifers, grazing has had minimal impact on the groundwater with nitrate concentrations averaging less than 2 mg/L.

Both piggeries and poultry activities have had significant impacts on the groundwater. Nitrate concentrations associated with poultry production is typically greater than 10 mg/L with one value up to 42 mg/L NO₃ as N. Piggeries appear to mostly result in higher concentrations with a range of values from 24–52 mg/L NO₃ as N.

The distribution of high concentrations of nitrate across a wide range of industries suggest continuing potential for nitrate contamination.

There is understood to have been a change in management practice in the poultry industry in the area. This is likely to result in a reduction in nitrate loads to groundwater in the area although there has been no monitoring to establish the impacts.

Case Study 10: Narromine, NSW— Country town, rural cropping

Narromine township lies in the Macquarie Valley and is about 40 km west of Dubbo.

The study area is underlain by approximately 100 m thick alluvial deposits associated with Macquarie River. It comprises distinct clay materials which are interbedded with sand/gravel units up to 20 m thick. The alluvium overlies an Ordovician/ Silurian sequence of slate, quartzite, shale and sandstone. The main aquifers are sand/gravel units which occur at depths generally greater than 40 m.

This case study considers land use in a regional country town. Irrigation cropping occurs outside the town.

Narromine Shire Council has been regularly monitoring nitrate levels in town water supply bores since the early eighties. This has shown that nitrate levels in most of the town water supply bores fluctuate over the period showing no set pattern in change of levels. Nitrate levels remained below 10 mg/L of nitrate as N. However one bore which is located down gradient direction of the town has consistently recorded nitrate concentrations up to a maximum of 8.9 mg/L. Other bores even those upgradient of the town typically range from around 2 to 5 mg/L nitrate as N.

The source of the nitrate in the groundwater at Narromine is not clear although there are considered to be numerous potential sources associated with a country town such as agricultural chemicals, sewage effluent, leaking sewer mains, stormwater drainage, cattle yards, a garbage tip and a cemetery.

Jewell (1996) concluded that pumping from town water supply boreholes generate an extensive cone of depression. The capture zone of the town water supply extends beneath the entire urban area. He concluded that all of the possible sources identified by Harwood (1986) may be contributing to the problem. Slow leakage through the upper clay sequence may delay impact for many years. There are no data or any suggestion of migration rates to estimate this period.

A further example of issues related to regional and country urban centres occurs in south west Victoria at a number of towns such as Koroit, Mortlake and Warrnambool, where there is elevated NO_3 (up to 150 mg/L NO_3 as N) in the groundwater in both the town and in the rural areas surrounding the town. The preliminary data suggest the introduction of nitrate in the groundwater system from outside the town, while an additional load occurs from town activities such as septic tanks, urban drainage bores and effluent disposal. There has been no detailed investigation to differentiate the sources of the nitrate in these cases.

Key reference: Harwood, 1986; Jewell, 1996

Monitoring of nitrate in Australia

The nature of the nitrate concentration data

There is considerable variability in the nitrate concentrations within particular land uses and differing soil and groundwater conditions. There is no unique response of groundwater systems to particular land uses.

However it is evident that high concentrations of nitrate are recorded in many environments throughout Australia at levels exceeding the 10 mg/L NO₃ as N guidelines.

Assessment of the reliability of the data is essential in interpretation of the data. The implications of the current and future extent of nitrate contamination is hampered by limited data from long-term monitoring of nitrate levels.

Nitrate monitoring programs

There is no consistent program of nitrate monitoring of groundwater throughout most parts of Australia. The major sources of data are generally related to specific research and investigation projects. Typically these data sets are restricted to the length and scope of the project and these are the data sets which are used in the case studies in this project.

A key source of nitrate concentration data is the one-off analysis of groundwater samples obtained from newly constructed bores. Although not providing an indication of any temporal variability, this information continues to provide data on the spatial distribution of nitrate contamination. This is particularly the case for broad scale diffuse source monitoring. This contrasts with smaller scale point source or multiple point source plumes which can be more readily identified by more targeted observation bore networks.

In some states there is routine monitoring of town water supply bores for health purposes for nitrate and a range of other parameters. The length of record varies considerably although these data sets are a potential source of long-term trend information.

A concern with the urban water monitoring is that samples are more likely to be recovered from production wells which may provide an average water quality for the entire aquifer and not detect increases which may be occurring in an aquifer system which is layered or contains preferred flow pathways. Furthermore, these samples are often associated with a combination of sources including point source as well as diffuse sources so that differentiation between sources cannot be undertaken.

Some significant data sets are also related to point source contamination problems such as effluent disposal and storage sites for landfills. Depending on individual State regulatory policies on dissemination of monitoring data, this source of information is often unable to be accessed.

Data density

As the nitrate concentration map shows (Figure 2 on page 11) there is a patchy occurrence of high nitrate groundwater in some areas of Australia. Results from many sampling sites indicate that nitrate concentrations are below 10 mg/L (as N). However there are many areas where there is a high density of samples, while elsewhere sporadic data points do not provide the same reliability on the presence or absence of nitrate contamination. The lateral distribution of nitrate analyses is not always related to particular land uses or to a particular program designed to investigate nitrate or one of its surrogates.

Therefore it appears that the current understanding of the extent of nitrate contamination is partly controlled by the monitoring data which are mostly derived from irregular sampling events. The currently available data do not necessarily indicate the complete extent of nitrate contamination in Australia.

Further, for those detailed sampling sites which do exist, the sampling points tend to be focused on the immediate area surrounding specific sites. There is often little monitoring which might show the spatial variability of nitrate contamination, particularly in relation to the physical or chemical character of the groundwater system. There is generally an absence of multi-level bores to provide an indication of the vertical distribution and how much of an aquifer is being affected.

The current picture of the distribution of high nitrate concentrations in groundwater across Australia is therefore partly a result of selective data collection and sampling programs.

Length and timing of data record

The data from several case studies indicate that at any location there can be a significant fluctuation in the nitrate concentration in groundwater (eg. Jandakot in Figure 3). There are groundwater systems in which the data indicate there is a change in nitrate concentration from high to very low in a relatively short space of time. It is not clear from the data how much of the aquifer (in a vertical or lateral sense) has such highly varying concentrations of nitrate.

There are generally limited data available from most of the case studies under consistent sampling conditions to either confirm a trend or relate the observed fluctuation in nitrate concentrations to a particular factor such as recharge or a pulse of nitrate moving through to the groundwater system.

Time series data are usually inadequate to establish seasonal and other variability in groundwater nitrate contamination. This is particularly the case for rural areas outside key groundwater supply areas. Long-term monitoring for some urban water supply areas include a wide range of nitrate sources which distorts the long-term trends from diffuse sources.

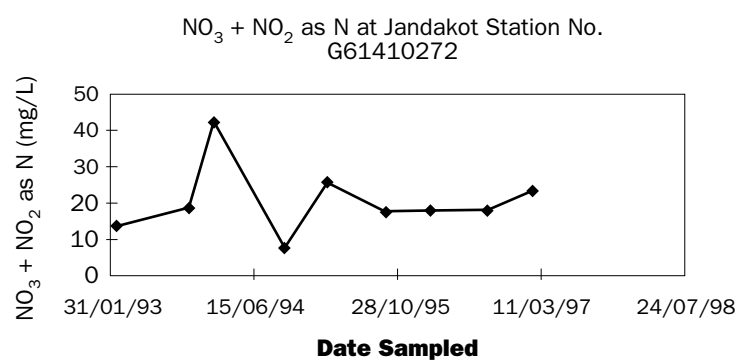
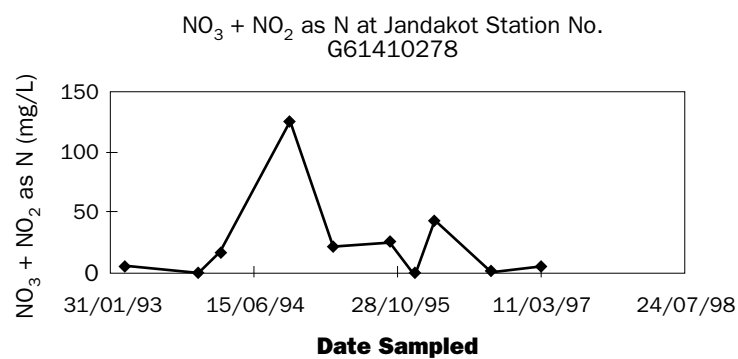
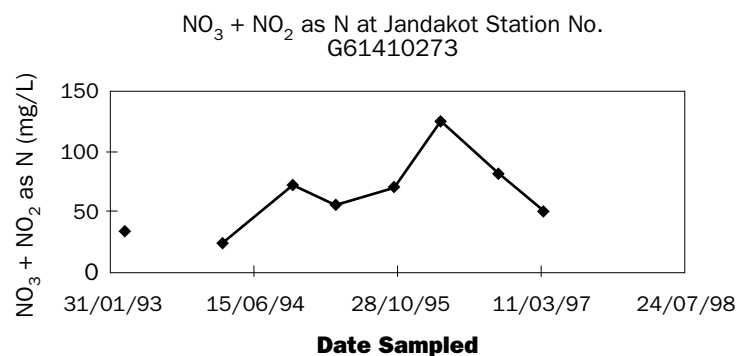
The data set for most case studies and from the State agencies data bases map indicates that there is no systematic approach to data collection of nitrate across Australia. Several of the case studies (eg. Perth, SE South Aust and Bundaberg) contain some detailed and long-term collection of data while most others rely on individual samples or non-systematic sample collection. Even in the case study areas there is little time series data which allows a satisfactory evaluation of the variability with groundwater recharge or other seasonal factors.

The absence of regularly collected long-term data makes it difficult to determine long-term patterns in nitrate contamination and does not allow adequate interpretation of the rate of any increase or decrease in nitrate concentrations.

Nitrate analytical data

Available data on nitrate concentrations are generally collected on an ad-hoc basis with no systematic sample collection and analysis program. There is added uncertainty as to individual sample recovery methods, preservation, laboratory procedures and reporting of the results.

Figure 3
Fluctuations of nitrate-N with time in groundwater at Jandakot, WA



Analytical procedures

Over the last 30 years analytical methods used for the detection of nitrate have changed significantly. Three major methods have been used by laboratories. These are listed below along with the shortcomings of each technique.

Ultraviolet spectrophotometric screening method

This technique was used to analyse for nitrate in the 1960s. Currently it is used only as a screening procedure due to interference by organic matter, surfactants, nitrite and Cr(VI) ions in samples. It can only be used reliably for uncontaminated natural waters, potable water supplies or waters with low organic carbon contents.

Results from the 1960s and 1970s that were obtained using this technique, may underestimate the nitrate present in the samples.

Nitrate ion selective electrode

This assay was developed in the early 1970s and used routinely by laboratories in the 1970s. The electrode is able to detect nitrate concentrations in the range 0.14–1,400 mg/L. The electrode has the disadvantage of functioning erratically when the pH of the sample is not kept stable. Unstable pH can result from the presence of humic or fulvic acids. To keep the solution stable, a buffer and a complexing agent are added.

Results from analyses using the nitrate ion selective electrode could underestimate or overestimate the nitrate concentration depending on the solution pH and what interferences were present.

Ion chromatography

This analytical method was developed in the mid 1970s and used routinely by laboratories by the mid 1980s. It is the most common method currently used for the detection of nitrate in waters. The method is, however, still subject to interferences. Co-eluting contaminants and contaminated glassware present the major problems.

Co-eluting peaks can result in the overestimation of the presence of nitrate in the sample.

Reporting and storage of analytical results

The reporting of the analytical results is not necessarily consistent, either between States, or within the time frame represented by the range of nitrate measurements. Of greatest significance is the reporting of the value of NO_3 as either N, NO_3 or NO_x . For example, in Victoria, early data (pre 1983) was expressed as nitrate and nitrite, while between 1983 and 1989, many analyses were reported as N but later sorted in the data base and recalculated to NO_3 (Shugg, 1997). Likewise, it needs to be clarified whether zero values mean that there is a true zero concentration rather than no sample analysis actually having been taken. Some rationalisation of data sets and establishing a clear set of data screening principles is needed.

Significance of nitrate contamination

Impacts of nitrate contamination on groundwater beneficial use

The beneficial uses of groundwater are a guide to the relative significance of the groundwater resource. There is a potential loss of the resource if the acceptable nitrate concentration for the particular beneficial use is exceeded. For example, if the nitrate concentration in potable quality groundwater exceeds 10 mg/L (as N), the groundwater is deemed to be unsuitable for drinking. Therefore the groundwater resource in this aquifer or part of the aquifer is lost for its particular beneficial use unless it is treated.

It is not realistic to attempt to place an economic value on lost resources resulting from nitrate contamination. This is particularly so due to the ongoing use of groundwater with nitrate concentrations exceeding 10 mg/L (as N). It is clear, however, that in areas of high nitrate concentrations, groundwater resources are unable to contribute to community water supplies and can result in the need for development of alternative and often less reliable supplies, often requiring expensive treatment schemes.

There are insufficient data available to assess the extent and volume of particular aquifers affected by high nitrate concentrations. However it is possible to make some generalisations concerning the impact of high nitrate concentrations on some aquifer types and aquifer systems and the current or beneficial use of the groundwater in some areas.

Aquifer systems known to be currently affected by nitrate contamination are indicated below:

- Basalt aquifers which are often associated with grazing, pasture development, local high intensity disposal of animal wastes and installation of septic tanks in urban areas appear to be susceptible to nitrate contamination. This is evident in the extensive basalt areas in south-west Victoria and around Toowoomba in south-east Queensland. This has led to the loss of the groundwater resource in a number of localities, including townships which have limited alternative water resources.
- There are similar concerns regarding the extensive grazing and dairying on the Limestone aquifers in south-east South Australia and a groundwater quality management plan is being implemented.
- There appears to be extensive nitrate contamination associated with irrigated pastures in a number of areas. This appears to have serious impacts on major potential uses of groundwater resources (such as town water supplies) and there are implications for the discharge of nitrate contaminated groundwater to ecological systems.
- Over application of fertilisers in sandy aquifers in the river deltas along the Queensland coast has resulted in groundwater in some areas of these systems from being unavailable to be used for domestic purposes.
- Over application of treated effluent and the waste disposal from septic tanks in coastal sandy zones (eg. Nepean Peninsula, Victoria) has resulted in the local groundwater being unsuitable for domestic use without treatment. In addition there are concerns of loss of the amenity of local beaches in locations where there has been excessive waste disposal.
- There are high concentrations of nitrate in aquifers throughout the Northern Territory. Although the preliminary evaluation suggests much of this nitrate is naturally occurring, it is unsuitable for direct use by remote communities. There is also some evidence pollution of local groundwater systems in remote parts of Northern Territory perhaps associated with the cattle watering at bore wellheads.

- The abundance of contaminated groundwater in urban areas from a variety of sources suggests that aquifers providing major parts of a town or city's water supply are in need of protection. Perth has been doing this in its major water source areas although there are still areas of contamination from local sources.
- It appears that there are numerous towns in which groundwater contamination by nitrate could be significant and result in planning restrictions, the need to develop alternative water supplies and expensive water treatments.

Characteristics of nitrate sources and impacts

Certain land uses clearly have an impact on groundwater resources. The major factors which affect the relative impact on groundwater contamination with nitrate are:

- the relative extent of the activity (eg. broad acre farming or high concentrations of point sources such as septic tanks);
- the approximate rate of leaching of nitrate from the land use activity;
- the ease of entry of nitrate into the aquifer (eg. gravity drainage through a fractured rock aquifer or slow leakage through a low permeability clay soil); and
- the aquifer conditions and the resulting effect of the different nitrogen sources on observed nitrate concentrations in groundwater.

These factors are indicated in the attached Table 4. This table is designed to indicate that for a particular source, whether nitrate will be readily introduced to an aquifer, if it is likely to be a localised input or a of much broader extent and what range of nitrate concentrations might result.

The table provides a method of assessing the relative significance of the nitrate source in relation to the concentrations generated and the area affected. Clearly, the broad scale activities typically have the potential to result in the widespread contamination of groundwater by nitrate.

Table 4
Characterisation of nitrate sources and impacts

Nitrate Source		Approximate Leaching Rate to Groundwater (kg N/ha/yr)	Area Likely to be Affected	Ease of Entry	Indicative GW Concentration Beneath Source (mg/L) NO ₃ as N*
(A) Broad Area Activities					
Dairy:	Irrigated	25 ⁽¹⁾	XXX	X–XX	>10
	Dryland	11 ⁽¹⁾			
Grazing:	Dryland	7 ⁽¹⁾	XXX	X–XX	>10
Fertiliser Application:	Market Gardens	130 ⁽⁴⁾ , 689 ⁽³⁾	XX–X	XX–X	—
	Potato/Dryland	57 ⁽⁴⁾			25
	Sugarcane	60–110 ⁽⁵⁾	(XX–XXX)	XX	10–15
	Viticulture	80 ⁽⁶⁾			—
	Field cropping	60 ⁽⁷⁾			—
Release of Soil N:			XXX–XX	X–XX	
	Clear felling	150 ⁽²⁾			
	Cultivation				>10 ⁽¹⁵⁾
(B) Multiple Point Sources					
<i>Intensive Animal Husbandry:</i>					
	Cattle Feedlots				
	Poultry Activities				
	Piggeries				
Septics Tanks:	Directly to sandy and rocky aquifers	6kg N/ system/year ⁽⁹⁾	X	XXX	47
<i>Garden and Lawn Fertiliser and Irrigation:</i>			X	XX–X	
	Public	30 ⁽¹⁰⁾			20
	Private	250 ⁽¹⁰⁾			20
Effluent Disposal:	Flood Irrigation	5 ⁽¹¹⁾	X?	XX?	45
	Potatoes irrigated with piggery effluent	200 ⁽¹²⁾			25
(C) Natural Sources			X–XX	XX	
	Termite Mounds				54 ⁽¹³⁾
	Mulga Woodland				10–20 ⁽¹²⁾
	Forests (pine)	4 ⁽²⁾			

Areal Distribution: XXX = widespread; XX = intermediate; X = localised.

Ease of Entry: XXX = direct input; XX = seepage through permeable medium; X = leakage through low permeability materials

Notes:

(1) Pakrou, 1997; Case Study 1.

(2) Smethurst and Nambiar, 1990; Case Study 1.

(3) Sharma and Aminuddin, 1996; Case Study 4.

(4) Pakrou *et al*, 1997; Case Study 1.

(5) Keating *et al*, 1996a, 1996b; Case Study 8.

(6) Juergens-Gschwind, 1989, in Schmidt *et al*, 1998; Case Study 1.

(7) Pakrou *et al*, 1997;

(8) DLWC (1992); Case Study 9.

(9) HydroTechnology, 1993; Case Study 3.

(10) Sharma *et al*, 1995; Case Study 6.

(11) DNRE, 1997b; Case Study 2.

(12) Dillon *et al*, 1997; Case Study 1.

(13) Barnes *et al*, 1992; Case Study 4.

(14) Schmidt *et al* 1998, Case Study 1.

* Source: As for leaching rate unless otherwise stated

Extent of the nitrogen-generating activity

Three overall categories of diffuse source nitrate contamination are identified which result in different areal extent of the nitrate sources in an area (Table 4). The extent of the sources result in the potential for different management practices to manage nitrate loads to the underlying aquifers:

- **Category A** activities including grazing, dairying and fertiliser applications have the potential to generate widespread nitrate loads. They tend to have limited regulatory controls and are geographically extensive. Not only is there potential for long-term degradation of groundwater water supplies, but ecological impacts on wetlands and base flow in streams at catchment scale. Of these activities, dairying is considered to be of most significance because it is least manageable at the surface. In contrast, application of fertiliser and release of soil nitrogen through cultivation (eg. fallowing of land) has greater potential for changes in land management.
- **Category B** activities such as animal husbandry, lawn fertiliser, effluent disposal and septic tanks, are significant sources of nitrate contamination. However, they are typically point sources which cause broad effects when aggregated. They therefore have the potential to be managed and often require regulatory controls.
- **Category C** sources are naturally occurring from a management perspective, groundwater resources affected by them may need to be considered for treatment rather than being concerned with the source itself.

Loading rate beneath the surface

There is a relationship between land use and applied loads. However load application rates coupled with the local conditions in the soil and groundwater means that the resultant groundwater concentrations will vary beneath areas of similar land use. Therefore the local site conditions need to be understood in order to determine the significance of the source on nitrate contamination.

The understanding of the key factors and quantification of the leachate generated is necessary to enable management of the surface conditions. This can result in minimising nitrogen loads, the generation of leachate and potential for nitrate migration to the watertable. The management practices adopted can influence the leaching rate and may vary from area to area for the same land use activity.

To illustrate this, Schmidt *et al.* (1998) in a study of the Mount Gambier region (included in Case Study 1), concluded that soil association and soil permeability were more important factors than land use in accounting for differences in groundwater nitrate concentrations. Other factors of importance included location, rainfall, depth to watertable, total bore depth and mean sampling depth.

Within each category in Table 4, indicative leaching rates below the root zone are given rather than the applied loads at the surface. This is because of the potential for the processes in the near surface soils to impact on the rate and concentration of leachate to the groundwater system. Some of the total loads can only be regarded as order-of magnitude estimates because accurate data on leaching rates beneath some land-uses are not available. Leaching rates for these land-uses had to be extrapolated from data for other land-uses or countries.

The case studies have indicated that the nitrate concentration in leachates migrating to the watertable are influenced by a range of local factors and processes:

- The soil zone plays an important role. This is due to the soil properties and also because of plants' ability to take up the nitrate. Plant nutrient requirements are often not balanced by nitrogen loads applied as fertiliser. This leads to generation of leachate for migration below the soil zone;
- Leachate can be generated from mobilising naturally occurring nitrogen by changing land use. This has been identified in Case Study 6 (Perth Metropolitan area) where there is evidence of mobilisation of soil nitrogen from *Acacia* country and in Case Study 1 where clear felling of pine plantations has occurred. Lawrence (1983) also demonstrated the change in nitrate concentration within a forested and unforested area. Careful management of land clearing is needed to minimise unnecessary introduction of nitrate to the groundwater system;
- Losses of nitrogen can occur in the surficial zone by volatilisation as shown in Case Study 2 for the effluent disposal at Werribee, potentially reducing the nitrate load leaching to the groundwater;
- Nitrogen can be lost where there are anaerobic conditions which could allow denitrification to occur. It has been postulated that denitrification occurs at some locations near Mt Gambier (Case Study 1) and in the Perth Metropolitan area (Case Study 6);
- Leachate generation is affected by infiltration and recharge rates. These conditions must be evaluated to manage nitrate concentrations and the extent of leaching. For example, areas of higher recharge but under the same loading (such as fertilised lawns in Perth, Case Study 6, Sharman and Aminuddin, 1996) have resulted in different nitrate concentrations in leachate below the soil zone which can then migrate to the watertable. In this case, though the loads leached were significantly different at particular times, the concentrations of leachate were very similar on an annual basis;
- High nutrient loads are generated from pasture (leguminous), from fertilisers and urination of grazing animals especially dairy cattle. This is strongly demonstrated by Case Study 1 in SE South Australia. Local factors in the soils can lead to variations in the rates of leachate generation and the concentrations differ even over small (paddock) scales;
- Direct application of nitrate loads occurs through sandy and rocky soil profiles as there is higher leakage including direct bypass of the soil zone. Pakrou *et al.* (1997) identified that preferential flow in basalts is a key mechanism for nitrate contamination of groundwater at cattle feedlots in Toowoomba; and
- Septic tanks represent a direct nitrate load. The leaching rate per hectare for septic systems is dependent on the density of septic tanks in the area. For example, ten systems per hectare will result in a relatively high leaching rate of 60 kg N/ha/yr.

Ease of entry into the aquifer

The characteristics of the unsaturated zone and watertable depth are basic controls on the ease of entry of leachate into the aquifer beneath a site or land use activity and may limit the rate of migration of the nitrate to the groundwater system.

For example rocky profiles (eg. Basaltic zones in western Victoria and SE South Australia, and in the Toowoomba area) as well as karstic environments allow high seepage rates through direct migration via gravity flow and result in elevated nitrate concentrations in aquifers

In contrast, there is evidence that in some zones where there is a low-permeability profile beneath the nitrate source, there is potential to minimise the *rate* of nitrate introduction to the watertable even though in such cases the concentration of solutes can be high.

Recent unsaturated zone modelling work by Sinclair Knight Merz (DNRE, 1997) on the concentrations of conservative solutes reaching the watertable from disposal of dairy waste supports the idea that there can be very long lag times in solutes reaching the watertable. Similarly, in the Perth area (Case Study 6), lower nitrate concentrations have been observed migrating through the more clayey Spearwood Sands than through the more permeable Bassendean Sands.

Aquifer conditions and observed nitrate concentrations

The concentration of contaminants reaching and entering the aquifer may be reduced by lateral groundwater flow within the saturated zone. This groundwater flow can reduce the potential nitrate impact on a receptor such as a production well or a discharge point at a stream or lake, and therefore reduce the significance of the nitrate contamination in the aquifer.

The key issue is the flux of nitrate entering the aquifer and whether there is any potential for degradation (by denitrification) or attenuation by dilution. The larger the thickness of aquifer within which mixing is assumed to take place, the greater is the groundwater throughflow at a particular location. Therefore there can be a greater degree of dilution of the nitrate in the aquifer and a reduced nitrate concentration in the groundwater.

Degradation of nitrate is not common once it is in the aquifer. However there is local evidence from the case studies, mostly in the Perth area, that denitrification occurs and that nitrate concentrations in groundwater are lower than would be expected if denitrification had not been occurring. The extent of denitrification in aquifers is not well understood.

Extensive plumes up to 4 km in length have been detected in aquifers into which waste has been injected (Shugg, 1993). There is evidence of degradation and denitrification in parts of the plume where the aquifer is anaerobic, but the nitrate is conserved where the aquifer is aerobic (Shugg 1993).

Vertical migration of nitrate within aquifers is common and this can distribute the nitrate load through an aquifer system. Most elevated nitrate occurs in the shallower (<30 m) intervals of aquifers. However data from Bundaberg (Case Study 8,) and from the Gambier Limestone in SE South Australia (Case Study 1) indicate that groundwaters sampled from more than 50m depth contain nitrate at elevated concentrations. Evidence of nitrate above 10 mg/L (as N) from aquifers in south west Victoria (Sinclair Knight Merz, 1996) also indicate vertical migration of nitrate within aquifers (Figure 4).

Not all concentrations at depth are necessarily above the guideline concentrations but they indicate the vertical migration of nitrate. This would be expected under conditions in which nitrate is conserved because there is no potential for denitrification. There is a need for additional monitoring of the depth profiles of nitrate in aquifers, coupled with evaluation of the vertical flow system and other physical properties of the system such as redox potential.

Observed nitrate concentrations for different nitrate sources

All of the land uses in the case studies, either individually or in combination, have resulted in significant concentrations of nitrate (>10 mg/L as N and occasionally above 100 mg/L).

High concentrations of nitrate in groundwater are associated with the following agricultural practices:

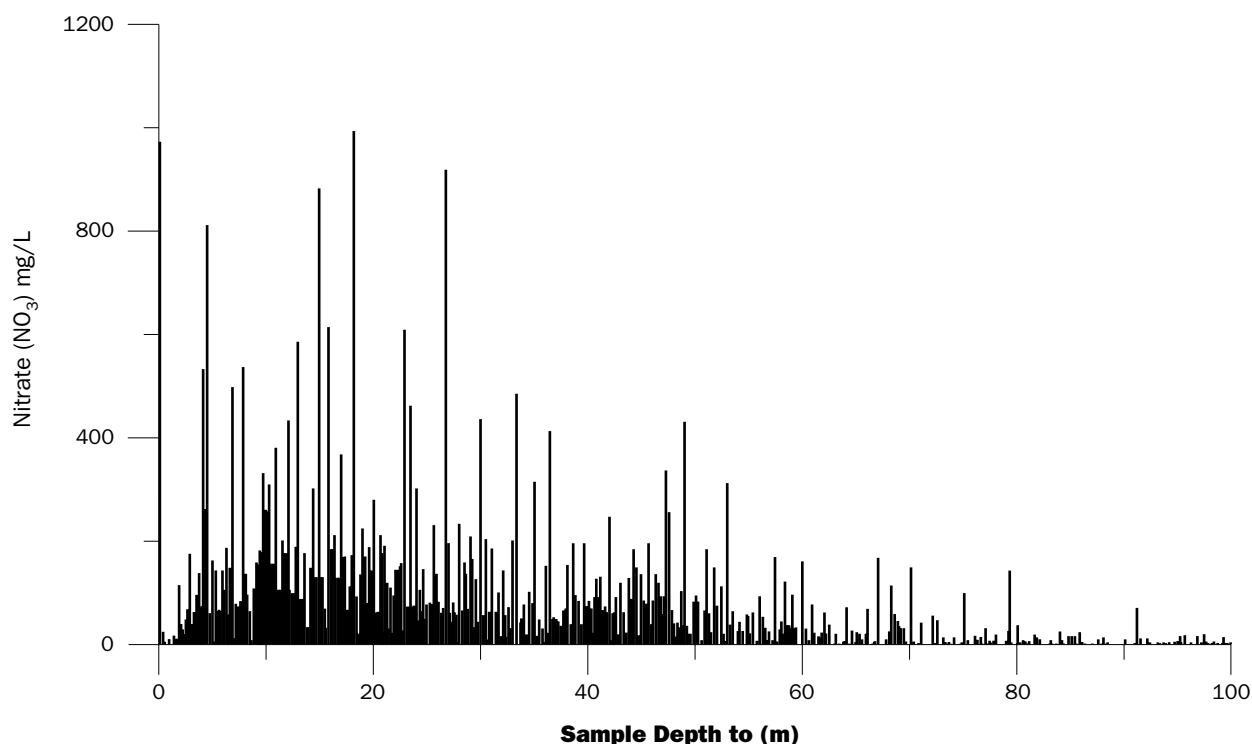
- High application of fertilisers in excess of plant nutrient demand, eg. Bundaberg, Perth;
- Over application of effluent above plant demand and soil moisture requirements. This includes waste from dairy, piggery and poultry operations, eg. Peel Valley, SE South Australia;

- Natural soil nitrogen mobilised by tillage and clearing of native bushland, eg. Perth area, Western Victoria (see Lawrence, 1983), Toowoomba;
- High nutrient loads through the soils from fertilisers on pasture, eg. SE South Australia,
- Nitrogen fixing pasture (often tilled and/or left fallow), often coupled with intensive grazing, eg. SE South Australia; and
- Over stocking or intense agriculture such as feedlotting, eg. Peel Valley, Toowoomba.

The following urban and rural residential developments are associated with elevated nitrate in groundwater in a number of areas:

- High concentrations are observed in all locations where there are septic tanks. This relates to both the density of the tanks, eg. Benalla, and the situation in which there is direct discharge of nitrate rich water to the unsaturated zone, eg. Nepean Peninsula, Perth. Only locally is there any opportunity for denitrification of the nitrate to minimise the groundwater contamination;
- Fertiliser applications at council and residential lawns and gardens, eg. Perth Metropolitan;
- Effluent disposal, although perhaps more of a point source than diffuse source, provides heavy loads of nitrate to groundwater, eg. Werribee, Wagga Wagga; and
- The combination of a range of undefined sources related to country towns; including the inflow of contaminants from rural activities outside the town limits, eg. Narromine, NSW and Mortlake, Koroit and Warrnambool in Western Victoria.

Figure 4
Nitrate concentration (mg/L) versus depth in southwest Victorian aquifers showing high concentrations more common at shallow (<30 m) depth, but significant numbers at greater depths



Potential for future nitrate contamination

Continuing sources of nitrate to aquifers

It is generally accepted that under most circumstances, once nitrate enters a groundwater system, its conservative nature ensures that it will remain in the system. This means that once the nitrate is in the system, there is a risk that the concentration of nitrate at a receptor such as a bore or a stream discharge point may exceed the acceptable concentration for the particular beneficial use of the groundwater.

Shallow, unconfined aquifers are generally at greatest risk of contamination from surface sources but there is potential for downwards migration. These aquifers are most susceptible to recharge processes and surface land use impacts. There is therefore a higher probability of nitrate accession to watertable than might be expected in deeper aquifers or those protected by some type of confining bed.

The time scale for migration through the soil zone, the unsaturated zone and within the aquifer varies depending on a number of factors. However there may be significant lead times between the initial release of the nitrate into the system and the accumulation of significant concentrations of nitrate in an aquifer or at a discharge point.

This phenomenon is shown in groundwater monitoring results from Europe and elsewhere where there is clear evidence that after prolonged periods of extensive fertiliser application groundwater became highly contaminated with nitrate after an extended lag period (Newson, 1994).

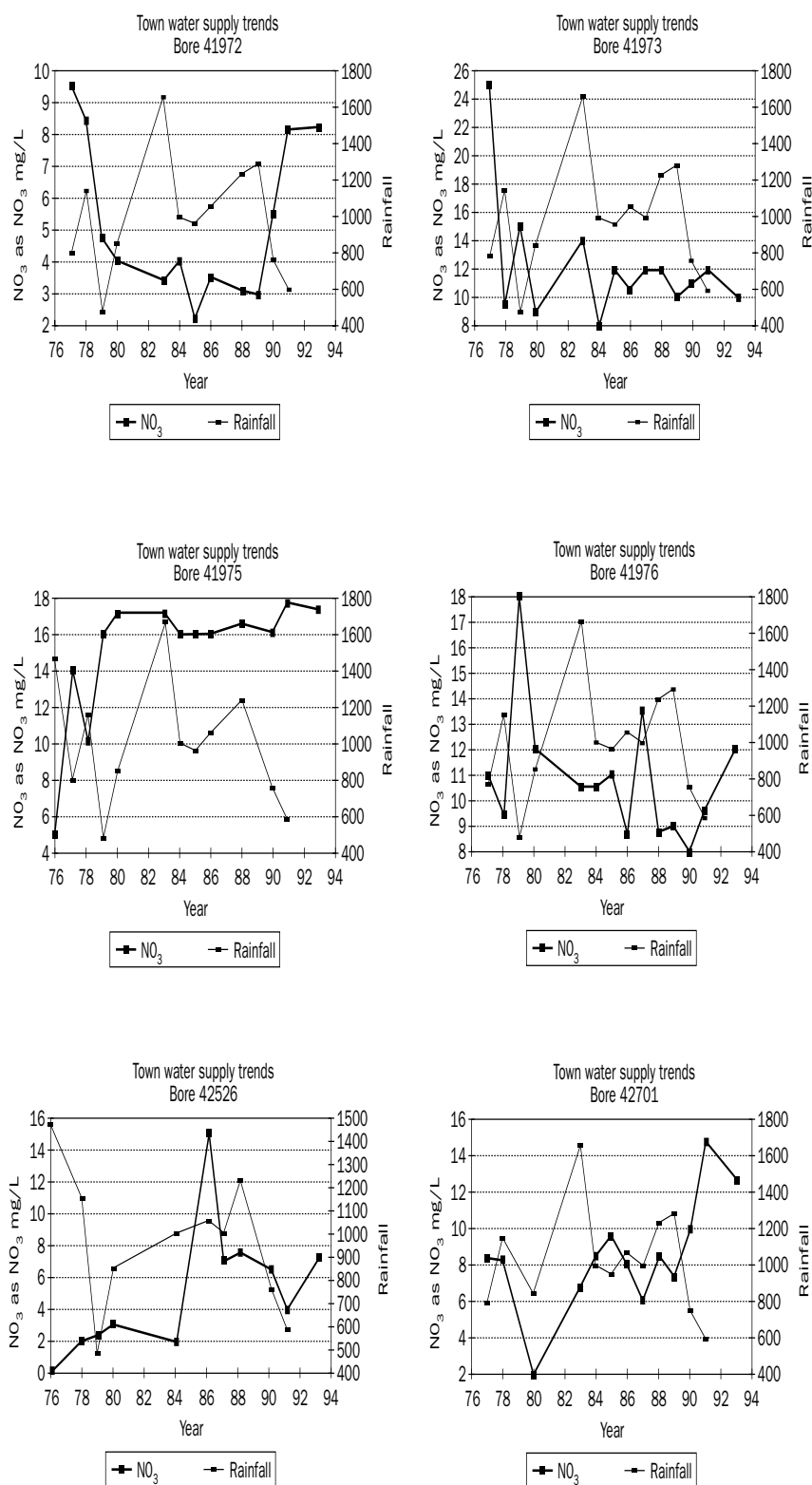
This lag period for nitrate contamination suggests that even though there are presently elevated concentrations of nitrate across Australia resulting from the impact of a wide range of human activities and natural processes, there is no certainty that the existing concentrations have reached their peak. This is supported by the modelling results of nitrate in aquifers in the southeast of South Australia reported by Dillon (1988). This suggests that there are possibly other regions where nitrate is stored in the unsaturated zone and slowly migrating to the watertable with the likely end result of elevated nitrate concentrations eventually being detected.

Furthermore, there is the potential for generation and migration of plumes of nitrate contaminated groundwater. To date the major investigation of plumes appears to be related to point sources such as injection bores at cheese factories (eg. Shugg, 1993).

The factors which influence the extent and rate of migration of resulting nitrate plumes are shown diagrammatically in Figure 5.

The figure assumes common application of a uniform nutrient flux into a uniform soil and groundwater system. It assumes that the ease of entry is greater in areas of high permeability soil and rock (eg. sandy soils in Bundaberg and Perth, rocky basalt areas such as Toowoomba, and deeply weathered granitic terrains such as those in the Peel Valley). In this situation, a dominant migration mechanism will be by direct advection and dispersion. In contrast, with low permeability zones, such as areas with deep dry profiles and low permeability metasediments (eg. Peel Valley), the ease of entry will be much less, with diffusion have a greater impact and advection will have lower significance.

Figure 5
Diagrammatic representation of nitrate concentrations in unsaturated and saturated zones



The figure indicates three main things:

1. It will take less time for the maximum soil concentration to be reached in areas of shallow watertable (ie. on the right of the figure). This means that areas of shallow watertable may have reached a maximum concentration and that ongoing application will lead to continued build-up of nitrate in the groundwater. In areas of deeper watertable there is likely to be a time lag.
2. The extent and distribution of nitrate in the groundwater plume is dependent on the aquifer permeability and the degree of mixing vertically. Continued input to the groundwater or the introduction of new loads (eg. in the left of the figure) will lead to an expanded plume(s). In areas of low permeability (and contaminant dispersivity), plumes may be very localised and concentrated. Extensive plumes may develop in areas of greater permeability and dispersivity.
3. The location of observation bores is critical in detecting the nitrate concentrations and the extent of plumes. In addition the selection of pumping wells for water supply need to consider the extent and behaviour of nitrate plumes.

Possible future directions for nitrate releases and loads for various land uses in Australia

For the land uses/nitrate sources identified, the case studies show that historically there is a high probability that nitrate will be released from the surface or subsurface source. ***It is therefore appropriate to assume that for any of the land uses identified, there is likely to be an ongoing release into the system, provided that the land use/management practice remains unchanged.***

The following discussion presents a judgement on the future directions of nitrate management which may influence the future risks of impacts from nitrate contamination in groundwater.

Risks from agricultural practices

The case studies have shown that there are significant risks of continued nitrate releases from the following agricultural practices:

- Fertilisers over-applied and in excess of plant nutrient demand;
- Continued grazing of pastures (eg. dairy cattle);
- Over application of effluent above plant demand and soil moisture requirements. This includes waste from dairy, piggery and poultry operations and from kennels;
- Changes to land use resulting in mobilisation of soil nitrogen by clearing of bushland and tillage of existing agricultural land; and
- Continued use of nitrogen fixing pasture.

These risks have the potential to be balanced by plant demand and thereby minimised, however there is unlikely to be sufficient incentive to farmers to change farming practices to allow this to happen. Under current practices, rates of application and nutrient loads are likely to be maintained. This will result in an increase in mass of contaminant into the system and potentially an increase in nitrate concentrations as recorded now in Europe.

In the event of expected increases in animal husbandry and feedlotting, increased environmental nitrate may result. Similarly, continued development of agri-products eg. fertiliser manufacture, abattoirs and dairy processing industries have the potential to result in higher nutrient loads discharged to water and soil environments. These operations will require substantive environmental management plans in the future to minimise waste production and disposal. These plans should also include minimising the opportunity for accidental releases through spills and overflow of effluent ponds.

The implementation of strict environmental management plans could be expected to decrease the current nitrate loads. For example, a poultry operation in the Peel Valley in NSW has recently adopted improved waste management. This adoption has included the implementation of sealed areas and collection of waste which was previously allowed to travel unregulated to the soil profile and adjacent streams.

Risks from increased land development

As the development of farmland and removal of native bush increases, there is increased potential for the mobilisation of soil nitrogen. This represents a potential for increased areal distribution of nitrate

Increased urban development could be expected to result in the following nitrate load issues:

- Increases in urban run-off and the load of nutrient and other waste;
- Increased number of septic tanks which have high nitrate loads;
- Increased hobby farms (rural residential) implying possible additional increased sewerage development for disposal—implies increase in sewage treatment plants and load, resulting in more land disposal;
- Solid waste disposal; and
- Manufacturing industry including dairy and poultry and piggery products.

Management of nitrate contamination in groundwater

Overall understanding of the nitrate problem

The case studies reviewed in this project have clearly indicated that in Australia a variety of land uses and situations in both urban and rural environments often result in high nitrate concentrations in the groundwater in the local area surrounding the particular land use. The impacts are mainly on the quality of groundwater resources and the base flow in streams. There is potential for the situation to become worse, as it has in parts of Europe, including the UK and elsewhere in the world.

However there is a range of technical problems and data deficiencies in relation to the understanding and management of nitrate contamination. This does not relate solely to identifying the current or future extent and the long-term trends in groundwater concentrations, but it relates to the recognition of the extent of major impacts, field extension and community education programs, planning of development, proposals and expenditure of research funds.

As a start to management of the issue of nitrate contamination in groundwater it needs be acknowledged that:

- There is a substantial amount of *general* information on nitrate contamination, including techniques to investigate concentrations, and plume extents, and to make predictions; and
- An integrated approach needs to be taken in key areas where nitrate in groundwater is a current or emerging problem. The local factors affecting load to the watertable and the processes operating need to be understood.

Technical issues required for management

The major technical issues relate to developing both an adequate data set to identify the spatial and temporal patterns of nitrate contamination and an understanding of the major processes affecting nitrate behaviour in the soil and groundwater system.

Data sets

The integrity of the data on nitrate concentrations is not consistent:

- Establishment of routine monitoring programs using consistent sampling periods, agreed sampling protocols and analyses, recovering samples from representative bores which provide a suitable aquifer thickness is necessary. Likewise it would be preferred to have consistent reporting of nitrate concentrations (preferably NO_3 as N mg/L);
- Consolidation of the data base on nitrate concentrations commenced for this study could provide greater focus on the key areas for future investigation and research; and
- The development of statistical techniques to refine the definition of nitrate source areas in localities where there is mixed land use and no clear relationship between nitrate concentration and land use.

As noted above, monitoring and generation of data sets need to be able to be interpreted on a sound understanding of the groundwater flow in the vicinity of a nitrate source so that the results of the monitoring and the impacts of the source on the groundwater system can be understood.

Nitrate loads and contamination processes

The importance of the local conditions on the ultimate nitrate concentrations in groundwater means that ***future works should be focused on areas at risk of groundwater contamination***. This requires identifying the areas to be targeted for detailed studies and improved management of various land use activities.

For each area at risk, the factors needing to be addressed relate to the nitrogen loading and the resultant leaching of nitrate to the watertable and the behaviour of nitrate within a groundwater plume. These are:

- For the soil and unsaturated zones, an integrated approach to land planning, both rural and urban, is needed from soil scientists, agronomists and hydrogeologists to develop techniques to minimise nitrate loads from particular activities. There is a need to understand the soil and unsaturated zone processes and the generation of nitrate leachate and its migration to the watertable.

There is a need to develop an understanding of:

- processes contributing to leaching for strategic land uses in risk areas and the effects of changes in land management practices on leaching;
 - the key drivers of nitrate loads through soil to the watertable and a quantitative assessment of leaching potential for different soils, land uses and management practices. This includes better understanding of nitrogen uptake by plants in particular soil and climatic regimes; and
 - the kinetics of nitrate migration through the unsaturated zone and the likely flux of nitrate to reach the watertable. This may include verification of existing and development of new modelling tools, including instrumented monitoring sites.
- In the saturated zone there is limited predictive capability for the longer-term attenuation processes within a groundwater plume.
 - groundwater samples from individual bores may record lower concentrations than others in the same area, which may reflect conditions in an aquifer system and the pathways by which nitrate may migrate within the aquifer system;
 - there is a need to fully understand the groundwater flow system in the vicinity of any of the types of nitrate sources so that the data generated on the nitrate loads and the monitoring that has been or may be introduced is able to be interpreted effectively;
 - it is generally assumed that once a nitrate plume is generated, it will continue to migrate downgradient toward a receiving environment with no loss of nitrate. From a site management perspective this can have implications for the siting of nitrate generating developments;
 - an understanding of nitrate dilution, dispersion and attenuation, as well as appropriate modelling techniques and monitored study sites, is needed. In addition data is required to identify the impacts on receiving waters and to assess the health risk which may result in potable aquifers with nitrate concentrations above drinking water standards; and
 - such an understanding will include detailed understanding of denitrification processes, and establishing guidelines for identifying sites where denitrification may occur.
 - The vertical extent of nitrate contamination within an aquifer is not well understood. More detailed documentation of vertical as well as lateral and temporal conditions would assist in understanding the distribution and potential migration of high nitrate groundwater within aquifer systems.

Leakage of nitrate through to deeper groundwater systems and the factors which may allow this to occur need to be considered.

- detailed understanding of the variation in groundwater nitrate concentrations, including the relationship of recharge to nitrate concentration is currently inadequate. This has implications in understanding the behaviour of nitrate plume migration and the ability to utilise nitrate-affected resources;
- in urban areas there is a need to identify the impact of sewers on nitrate sources in groundwater; and
- The distribution of nitrate contaminated groundwaters on receiving water environments, including the ratio of N:P, is necessary to establish the impact of discharge of nitrate contaminated groundwater and the siting of potentially contamination activities close to receiving waters.

Policy implications

The implementation of the technical programs outlined above will in some cases require or result in major education programs, improved planning practices and perhaps environmental protection regulations. The overall objective is to reduce accession of nitrate below the root zone by careful management.

Some general policy directions to achieve a reduction in the load of nitrate to groundwater and their implications are presented below.

Management of farm wastes

Farming practices which better manage animal waste loadings on paddocks will assist in minimising the load of nitrate to the watertable. More intensive animal husbandry requires well designed waste collection, management and effluent reuse schemes.

More efficient waste management could result in better economic returns in the longer term.

Urban development applications

New urban development proposals need to clearly establish the total nitrogen balance resulting from the development including household and industrial effluent, disposal of solid waste (a point source of nitrate), fertiliser applications to household and urban gardens. Appropriate development guidelines should be adopted and implemented by local planners.

In areas where there is a potable groundwater resource, the implications of drawdown of water levels and potential migration of nitrate contamination into the zone of groundwater extraction will need to be identified.

Similarly, the implications of developments on waterbodies need to be established.

Groundwater monitoring of urban supply bores

Routine monitoring of urban groundwater supply wells should be conducted and the analytical results made available on relevant State Groundwater data bases. Although this practice is conducted in some States this should be mandatory. The data should be regularly interpreted and reported.

In the event of major water resource nitrate concerns for health or environment, nitrate protection zones should be established. The criteria for assigning such a zone should be based on the beneficial use of the groundwater. Establishing the form of such a zone may be either site/area specific for major problem areas requiring specific investigations. In smaller zones, a series of generic criteria could be developed.

Monitoring of domestic water bores

All bores used for domestic purposes should be analysed for presence of nitrate on a regular basis and the results forwarded to a central State groundwater data base. This may require alteration of the bore classification in some areas which currently refer to stock and domestic bores.

Community participation in groundwater protection

Diffuse sources dominate nitrogen loads to aquifers and can realistically only be effectively managed by landowners who are aware of the potential for contamination and the means to reduce and/or prevent it.

Groundwater protection is within the issues addressed by the Landcare movement but received scant attention mainly due to the lack of available information on these factors, at a community level and in many cases, in resource management authorities. Knowledge generation and transfer are required.

Conclusions

Nature of the problem

Contamination of groundwater by nitrate in Australia is widespread and occurs over regional and local scales. Background nitrate concentrations are in the order of less than 2 mg/L NO_3 (as N). In many areas the concentration is greater than the recently revised *Australian Drinking Water Guidelines* (NHMRC–ARMCANZ, 1996) level of 50 mg/L nitrate as nitrate (11.3 mg/L as N) recommended drinking water quality guideline level of 10 mg/L and makes the groundwater resource in these areas unfit for drinking. In some of the more contaminated areas the concentrations exceed 100 mg/L.

As well as impact on direct use of groundwater, high nitrate concentrations in groundwater discharge affect the water quality in receiving environments leading to eutrophication and to development of algal blooms.

Data limitations

The understanding of nitrate contamination and the processes causing it has been developed in Australia largely from a randomly collected set of data. Some targeted studies have been undertaken over limited time frames or in relatively small areas, and provide only a snapshot of information or a localised evaluation of the nitrate contamination. There is limited routine monitoring of nitrate in groundwater and there are numerous uncertainties regarding the nature of the data available. Time series data are typically inadequate to establish seasonal and other variability in groundwater nitrate contamination. Many of the nitrate analyses available have been obtained from water samples taken at the initial construction of a bore.

There are also concerns regarding the quality of the early analyses and the manner in which results are reported. There is still no consistent way of reporting nitrate concentration although most workers report concentrations as NO_3 as N rather than NO_3 as NO_3 .

Nature and extent of nitrate sources

Nitrate contamination of groundwater is associated with a wide range of sources of nitrogen which occur in both rural and urban environments.

The extent of the nitrate contamination depends on the nature of the source. Improved management of nitrogen sources could reduce the ongoing release of nitrate into the groundwater system. Many of the nitrogen sources have been managed in such a way that excessive loads of nitrate have migrated below the soil zone and reached underlying aquifers.

Nitrate loads from activities such as grazing or clear felling have the potential to result in broad scale release of nitrate resulting in areally extensive impacts on aquifer water quality but limited opportunity for improved management. In contrast application of fertilisers for cropping or pasture could be managed by limiting applications to meet plant requirements.

Point sources or multiple point sources are more localised and have potential for reduction in nitrate loads to groundwater. Developments such as manufacturing and processing of agricultural products, solid waste and effluent disposal typically require regulatory approval and restrictions are able to reduce the actual load of nitrate for disposal.

Potential future impacts

Current management practices have not recognised the implications and complexities of nitrate contamination and the means of minimising nitrate inputs to groundwater.

There is a significant probability that with a continuation of current farming practices in Australia there is unlikely to be a significant reduction in environmental nitrate loads available to migrate to the watertable. The ongoing clearing of land for both urban and rural development represents a potentially increasing source of nitrates from a variety of both diffuse and point sources.

In addition under particular circumstances, a time lag exists between the surface release of nitrate and entry into the aquifer. Therefore there is potential that nitrate stored in the unsaturated zone has not yet reached the watertable and currently unaffected aquifers are at risk of contamination or increased concentrations could result.

Understanding of risks and processes

There is wide variability in the conditions which yield high nitrate concentrations in the groundwater. The highest concentrations are found in the shallow unconfined aquifers which are most susceptible to contamination, but migration to depths of 50 m or more does occur. The potential layering of nitrate concentrations means that in areas of thick aquifers multiple observation bores are likely to be required.

It appears that where a source of nitrogen exists there is the potential for nitrate to reach the groundwater beneath the source. However the concentration of leachate which reaches the groundwater depends on local conditions at the source. ***That is, certain areas are at greater risk of being contaminated by nitrate than others.***

Regardless of the natural conditions at a source of nitrate contamination the most appropriate means of minimising nitrate loads to groundwater is to carefully manage the application rates taking the local site factors into account.

The key factors which influence the ultimate load to the groundwater are:

- whether nitrogen can be used effectively at the surface by plants in the soil zone or there can be minimal leakage below the soil zone (particularly in the case of a point source);
- variations in soil type and recharge rate which controls the rate of leachate migration through the soil zone;
- the conditions in the soil zone and the unsaturated zone can prevent the production of nitrate or denitrify nitrate; and
- the conditions in the aquifer allow denitrification or attenuation can occur by other means such as dilution.

In general there is little detailed understanding of the groundwater conditions and the nitrate contamination processes, so that the interpretation of the limited data is likely to be inconclusive. This is both at local and regional scale. It is important that, particularly in risk areas, the characteristics of the groundwater system including the soil and unsaturated zones, are carefully understood to enable an evaluation of nitrate contamination to be undertaken.

Future investigations and research should focus on obtaining a clearer understanding of the nature and extent of nitrate contamination. The key issues are to:

- develop a comprehensive set of reliable data on which to base interpretations and conclusions;
- undertake detailed research of the key processes of nitrate contamination to obtain greater understanding of the relationship between nitrate release and downwards migration and resulting groundwater contamination and impact; and
- investigate policy issues for changes to land and water management, including dissemination of information regulators, farmers, planners and developers.

Recommendations

This study has identified a broad range of issues associated with the existing and future management of nitrate in groundwater in Australia. The following recommendations are put forward to establish management actions, and to reduce gaps in our existing knowledge.

Management and policy development

- Change the focus of policy and research on nitrate contamination in groundwater from point sources of nutrients (which can be managed) to broad area diffuse sources (which require more complex management);
- Develop guidelines for groundwater protection zones around major potable water supply areas specifically focusing on nitrate sources;
- Liaise with State Agencies to develop programs for land management which improve nutrient applications in broad area farming; and
- Encourage the COAG Water Reform Committee to more actively include water quality (particularly nitrate) in developing future policy decisions.

Confirmation of nitrate trends

- Develop suitable groundwater monitoring networks in key areas where nitrate contamination from existing land use is already known. This will need to be conducted in association with State Agencies. Typical priority areas based on the Nitrate Map prepared for this study include Toowoomba, Perth coastal plain, broad area grazing country in Victoria, NSW and SE South Australia;
- An emphasis should be placed on fully categorising the extent of nitrate pollution laterally and vertically; and
- Evaluate any long-term urban water supply monitoring data on nitrate concentrations to establish nitrate trends. This will incorporate information in many areas around the country which are outside the study areas in this project.

Research activities

- Establish techniques for identifying risk to groundwater from nitrate pollution around key industries which can produce high nutrient loads. Two major tasks are proposed:
 1. Risk mapping in areas where there is broad area nutrient loading; and
 2. Development of a 'pollution index' for more localised nitrate source management, eg. point sources, taking into consideration factors such as watertable depth, lithology, groundwater flow system and groundwater beneficial use.

Priority areas for these activities are:

- areas of dairying;
- areas of high application of fertilisers for horticulture over broad areas;
- areas where there are urban water supplies; and
- high-value groundwater sourced waterbodies and ecosystems.

- Establish research into denitrification of nitrate, including determination of the key processes and conditions allowing denitrification. The relationship of denitrification to climate is an important issue.
- Establish trial sites in areas where land management is changing and monitor impacts on nutrients in both soils and groundwater, eg. areas in SW Victoria which are soon to be or have recently been changed from forested to agricultural land use.

Funding allocations

As a guide to LWRRDC on the way in which it should meet these recommendations, the following allocation of responsibilities for funding and the priority for each task is presented in Table 5.

Table 5
Responsibilities and priorities for future works

Item	Priority for LWRRDC	LWRRDC %	State Agencies %
Monitoring	5	5	95
Research			
Risk Mapping	2	30	70
Denitrification Studies	3	100	
Establishment of Trial Sites	4	40	60
Management and Policy			
Develop Protection Guidelines	1	50	50

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Case studies

Pasture, mixed agriculture and forestry—south-east of South Australia

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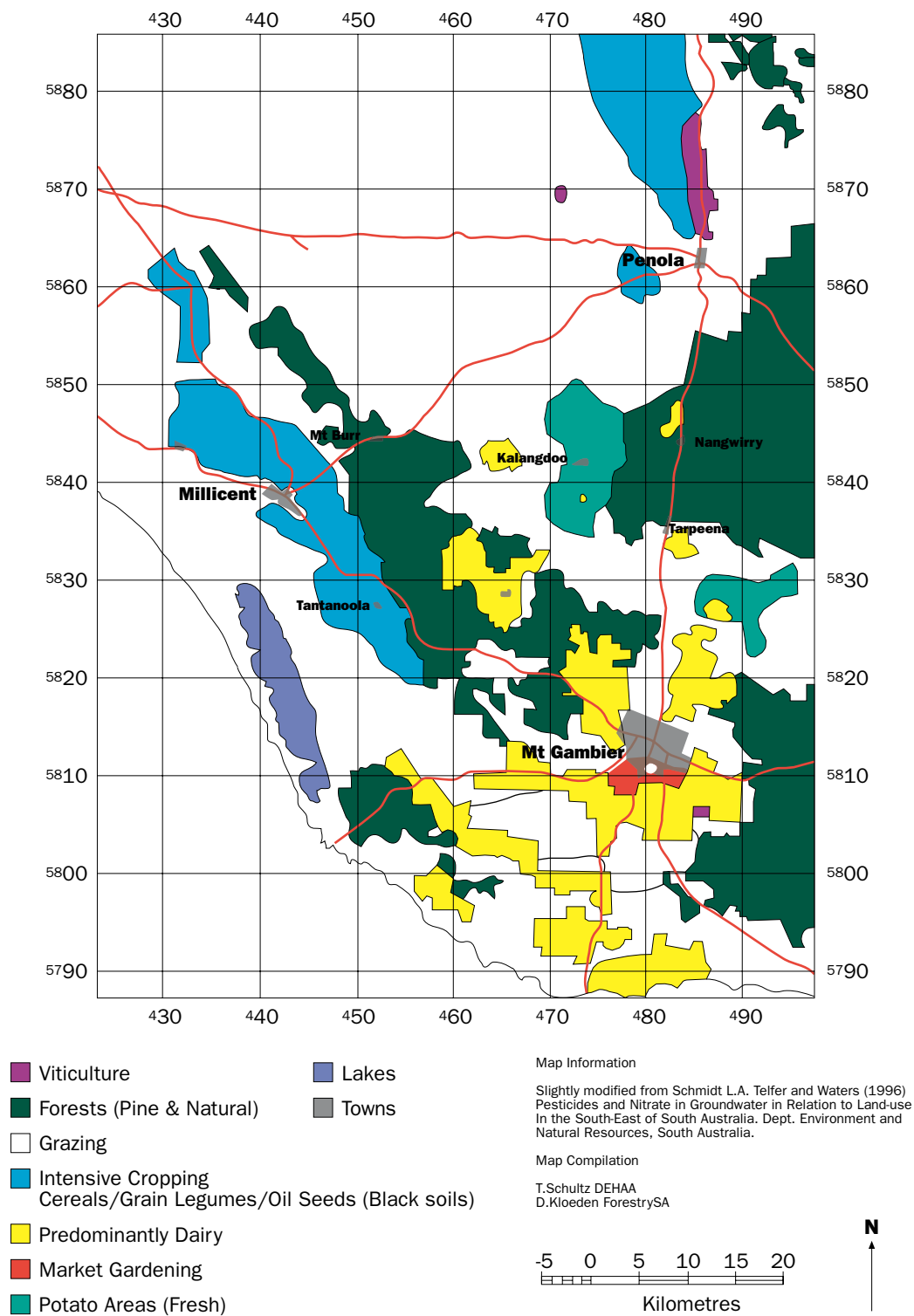
Previous studies, area description and land use

The Gambier Plain of the south-east of South Australia and western Victoria has been the site of a series of investigations over 25 years on nitrate concentrations in groundwater, which have aimed at identifying and quantifying the causes of nitrate pollution. These have enabled ranking of the relative importance of the various sources, and to formulating management strategies to more effectively protect groundwater quality. Nitrate is not the only groundwater pollutant in this area, but it is the most common.

Table 6
History of studies of recharge and nitrate concentrations in groundwater in the south-east of South Australia

Authors	Topics
Recharge	
Holmes and Colville (1970; 1970b)	grassland and forest hydrology
Allison and Hughes (1978)	recharge estimation by chloride balance and tritium methods
Nitrate	
Waterhouse (1977)	hydrogeology, and nitrate concentrations in groundwater
Harvey (1979)	nitrate concentrations in groundwater, and nitrogen loading rates
Schrale and Magarey (1979)	criteria for land treatment of cheese factory wastewater
Lawrence (1983)	nitrate rich groundwaters of Australia
Ockenden (1985)	south east region water resources management review
Dillon (1988)	causes of nitrate contamination near Mount Gambier
Smethurst and Nambiar (1990)	nitrogen balance in pine plantations
Richardson (1991)	fate of cheese factory and abattoir effluent
Love, Armstrong & Stadter (1992)	Otway Basin hydrogeological investigation
Bauld, Sandstrom and Stadter (1995)	groundwater quality—Padthaway—Coonawarra
Barber (1995)	aquifer vulnerability mapping
Dillon, Schrale and Deinum (1995)	groundwater quality at effluent irrigation sites
Telfer (1992)	Blue Lake water quality management plan
Schmidt, Telfer and Waters (1996)	pesticides and nitrate in relation to landuse
Pakrou (1997)	impact of grazed dairy pastures on nitrate in groundwater
Dillon <i>et al.</i> (1997)	monitoring leachate beneath piggery effluent irrigation area
Schmidt, Schultz, & Schrale (1997)	nitrate pollution in relation to land management systems

Figure 6
Land use for Mt. Gambier study area



The report by Schmidt *et al.* (1997) draws together all of the existing nitrate data, and hydrogeological and land use information, for an area of approx 6,000 km², a 70 km wide strip bounded in the east by the South Australian–Victorian border, and running from the Southern Ocean in the south to Coonawarra in the north. The area is part of the Gambier Plain, and has low relief with occasional stranded coastal ridges (Bridgewater Formation) parallel to the coast, and some isolated volcanic cones, most notably mount Gambier and Mount Schank.

The area is underlain by Gambier Limestone, a Tertiary deposit which increases in thickness towards the coast reaching up to 350 m thick. There is extensive karst development within the limestone, particularly in the vicinity of the watertable. However the karst features are not connected in any laterally extensive system, and groundwater flow occurs primarily as intergranular flow. Underlying the Limestone is the Dilwyn Formation, a groundwater system confined and separated from the limestone by clays.

The soils in the study area have been formed by weathering of parent limestone, and by aeolian transport of sands and clays. The other main class of soil is volcanic. Soils derived from weathering are generally shallower than 1 metre. Most of these soils are relatively permeable, although there are swampy areas where there are surficial clays or impeding clay layers at depth.

Mean annual rainfall in the area varies from 650 mm to 800 mm and average annual pan evaporation is 1,300 mm to 1,500 mm. The climate is Mediterranean, with cool wet winters and warm dry summers. Mean monthly temperatures range from 9° to 18°C. In an average year monthly rainfall exceeds pan evaporation for four months (May to August).

The watertable is shallow, ranging from depths less than 2 m to more than 20 m in the Mount Gambier area. Groundwater flows generally towards the coast, and typical velocities are 5 to 50 m/year (Love *et al.*, 1992). Recharge rates are typically from 30 to 300 mm/year in the permeable soils (Allison and Hughes, 1978), but this also depends on land use. For example recharge rates in pine forests on sandy soils are negligible after canopy closure (Holmes and Colville, 1970).

The current distribution of land uses is shown in Figure 6. Clearing of native forests began around Mount Gambier in the 1840s. Land clearing intensified in the late 19th century, and today over 90% of the original vegetation has been replaced by primary production. The dominant land use is dryland grazing (66% of the area), followed by forestry (20%), field cropping (7%), and dairying (4%), with the remainder consisting of horticulture, viticulture and urban and industrial development.

In the last decade or so, the proportion of dairy pastures which are irrigated has increased. Another development has been an increase in potato growing, generally as a one in four year rotation with dryland grazing, and the area covered is now approaching 2% of the total area.

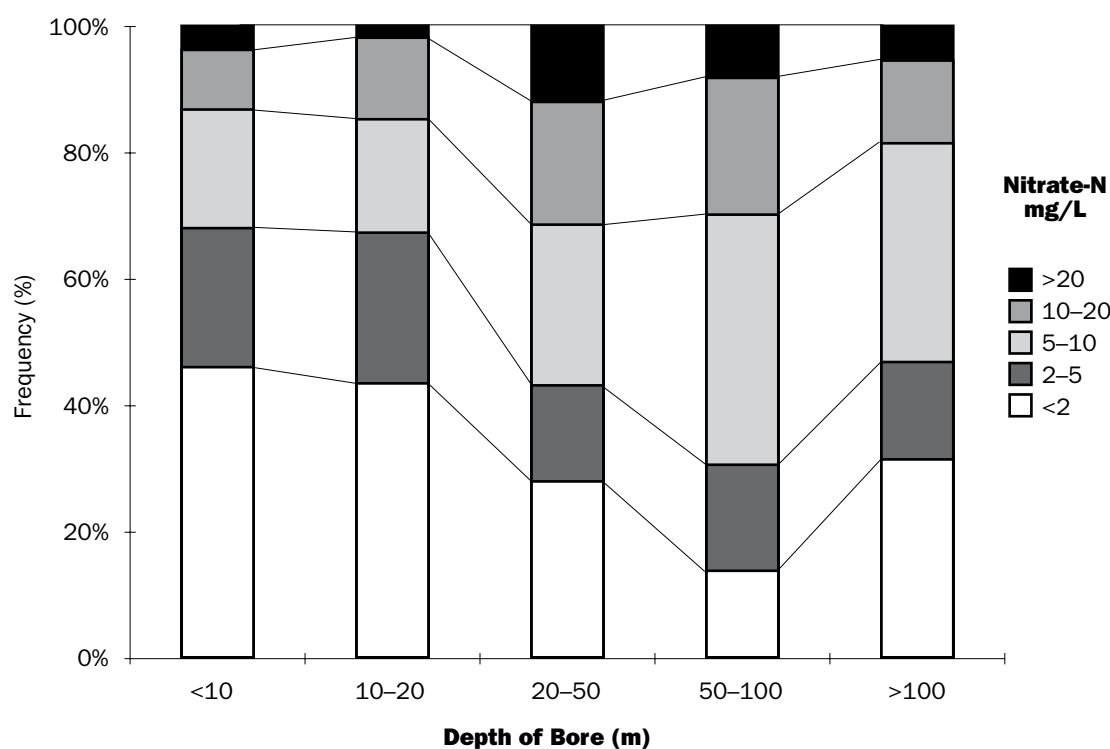
A study of the most intensively developed part of the area, 1,000 km² in the south eastern corner of the study area, revealed more than 300 point sources of nitrogen (Dillon, 1988). This included 201 dairies, 39 abattoirs and saleyards, 21 dairy factories, 12 piggeries, and 11 solid waste disposal sites. Most of the abattoirs and dairy factories had closed due to consolidation of primary industry processing over more than 50 years. Sewage effluent from the major towns in the area is now collected into common effluent schemes, and disposed either via treatment plants to the Southern Ocean (Mount Gambier, population 22,000), or to land-treatment (Millicent, pop 5,100; Penola, pop. 1,200). Smaller towns rely on individual septic systems, and for some of these, there are plans to establish common effluent schemes.

Nitrate concentrations in groundwater

Schmidt *et al.* (1997) report three studies on nitrate concentrations of bores in this area, with a total of 1,663 observation locations. Due to the different sampling strategies and overlapping but not coincident geographic areas covered, the percentage of wells reported as exceeding a nitrate concentration of 10 mg N/L ranged from 11% to 33%, and for the combined data was 24%.

Statistical analyses were performed to determine the relationship between nitrate concentration and depth of sample extraction as inferred from a range of bore parameters; total depth, depth to the top of the extraction interval (maximum of depth to watertable and depth to base of casing), the extraction interval, and the mean sampling depth. Most wells are completed as open holes, with casing only in the upper part of the hole down to the Gambier limestone. The correlation analyses indicated no significant ($p > 0.05$) linear relationship between nitrate concentrations and any of these parameters. However it is apparent that the highest proportion of bores with nitrate concentrations exceeding 10 mg N/L occur when the top of the sampling interval is less than 50 m deep (Schmidt *et al.*, 1997). The frequency of high nitrate detection as a function of bore depth is shown in Figure 7.

Figure 7
Nitrate concentration at various bore depths



Regression analyses were also carried out to determine the proportion of variance between samples which they could explain. The parameters considered, the variance they explained and their significance are shown in Table 6. Although a number of the variables were significant, only soil association and soil permeability individually explained more than 10% of the variation. On its own land use explained only 10% of the variation. A multiple regression model showed there was a strong correlation between soil association (29 degrees of freedom) and soil permeability (one degree of freedom) and adding permeability to land use increased the variance accounted for to 29.5%, compared with land use and soil association (36%). Soil permeability is therefore a good surrogate for soil association in describing variations in nitrate concentrations in groundwater.

It was found that the differences in nitrate concentrations for the three data sets were insignificant once account had been taken of the geographic extent of each study, land use and soil association. That is the three data sets were found to belong to the same population, justifying the analysis of all the data together.

Temporal variation of nitrate concentrations was addressed in Dillon (1988) in two ways. Firstly a set of 42 bores had been sampled in 1972 and 1982, at the same time of year, using exactly the same sampling procedure. The mean nitrate concentration increased from 16.7 to 16.9 mg N/L but the difference was statistically insignificant. Secondly, an analysis of four sets of bores, each with a time series of samples, gave rise to semi-variograms of nitrate concentration variance with respect to both time and space. In all cases there was significant autocorrelation. Autocorrelation coefficient exceeded 0.9 for 10 years for regional observation wells, and for 2 years and 1.4 years for bore networks used to monitor point sources of pollution.

Therefore aggregating data over 25 yrs in Schmidt *et al.* (1997) is known to result in some variance purely as a result of temporal variations. However the magnitude of this unaccounted affect is expected to be very small with respect to other causes of variation.

Table 7
Percent variation of nitrate-N concentrations explained by different variables, determined by regression analysis

Variable	Degrees of Freedom	% Variation Explained (A)	Significance of A
Soil association	29	31.7	P<0.001
Soil permeability	1	19.2	P<0.001
Land-use	7	10.0	P<0.001
Data set	2	7.4	P<0.001
Grid reference	2	6.0	P<0.001
Rainfall	1	2.8	P<0.001
Depth to water	1	2.1	P<0.001
Mean sampling depth	1	1.9	P<0.001
Total bore depth	1	1.0	P<0.001
Extraction interval	1	0.2	n.s.d
Top of interval	1	<0.2	n.s.d

A = R² adjusted for degrees of freedom

n.s.d = not significantly different

Attempts have been made to partition nitrate concentration data into two sets consisting of those likely to be affected by point sources of pollution, and those unlikely to be so affected. Waterhouse (1977) and Dillon (1988) have used various screens, such as proximity to known pollutant sources to sort the data. In the latter case a 'plume filter' with a length of 4 km in the direction of groundwater flow and a width of 1 km, maximised the significance of the difference in the means of the discriminated data sets.

However, when used as a predictor of kriged block mean nitrate concentrations in a multiple linear regression, the fraction of samples in the block which are affected by known point sources (within plumes of this dimension) accounted for only 7% of the variance between concentrations. Obviously factors such as strength of the source and length of operation could also be taken into account, but based on Dillon's (1988) analysis, point sources contribute only a small fraction (11%) of the nitrogen load to the aquifer and further refinement of partitioning methods is unlikely to yield a greater ability to account for the variance in nitrate concentrations in the aquifer.

The low proportion of variance explained by the regression against physiographic and land-use factors is not surprising, as there are significant differences in land use histories within any current class of land use. Within any soil association, there are also large variations at a small spatial scale. There are also significant differences in land management practices within any land use. This last source of variance, is both problematic to define and is also a sign of promise that within the range of current management practices, there are some which give adequate protection of groundwater quality.

Processes leading to leaching of nitrate to groundwater

Leaching of nitrate requires nitrate to be present in the soil at a time when recharge occurs. The dominant sources of nitrate are different in different land uses. Recharge rates can also vary depending on the uptake of water from the soil by plants over the wetter times of year, and the rooting depth of the vegetation.

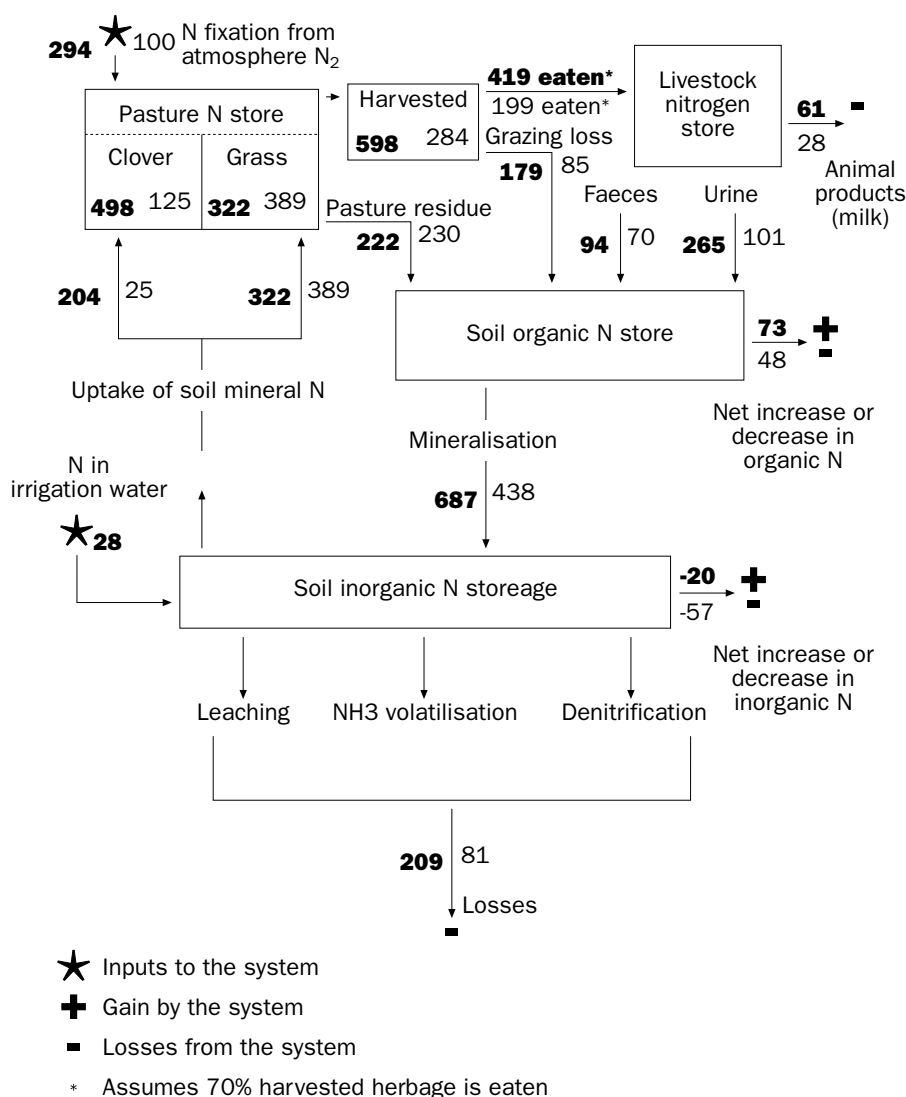
Dryland and irrigated dairy pastures

A comprehensive study of the nitrogen and water balances in dryland and irrigated dairy pastures has recently been completed by Pakrou (1997). This found that dairy cow urine is the primary source of nitrate leached to groundwater. This nitrogen originates from atmospheric nitrogen fixed in the pasture by microorganisms associated with clovers which are pasture legumes. There is also substantial recycling of soil organic nitrogen by mineralisation, making available inorganic nitrogen which can be taken up by the pasture to become feed, and that which is not taken up or immobilised is susceptible to leaching. Gaseous losses of nitrogen are only a small component of the nitrogen cycle in these pastures.

The nitrogen contained in milk is only about 14% of the nitrogen consumed, and the nitrogen turn over through livestock wastes is significant, especially where pastures are irrigated and have higher stocking rates. Irrigated pastures also have summer-active grasses which are dormant in winter in comparison with species in dryland pastures. The reduced transpiration and nitrogen uptake in winter, when water drains through the profile also contributes to higher leaching losses of nitrate to groundwater. The higher moisture contents in autumn in irrigated pastures also lead to earlier leaching of water through the root zone and greater volumes of leachate than in dryland pastures. Figure 8 shows the nitrogen balance for dryland and irrigated pastures with total losses of 80 and 210 kg N/ha/yr respectively of which approximately 10 and 25 kg N/ha/yr are shown to be leached. The study was undertaken at OB Flat, about 10 km south of Mount Gambier over the period 1991–1995. Nitrate concentrations were approx. 10 mg N/L in groundwater at this site, where watertable depth was 21 m. A summary of research outcomes is provided in Pakrou *et al.* (1997).

Figure 8

Nitrogen flows and balance in irrigated (bold) and non-irrigated paddocks



Source: Pakrou, 1997

Dairy sheds are a point source of effluent with potential to pollute aquifers. A successful community awareness and education program has been run with the Dairy Effluent Guidelines Group to develop and implement guidelines to manage dairy shed wastes. This has been reinforced with an Evangeolu environment protection policy dealing specifically with dairy shed effluent. This is seen as a model to involve other groups in direct action by the community or industry to reduce or eliminate major collective point sources of nitrate to groundwater.

Potatoes irrigated with piggery effluent

A study of the leaching of nitrogen and salt from a potato field irrigated in part with piggery effluent for one season in a four year rotation in a dryland pasture was performed in 1994–1995 near Mingbool, 25 km north-east of Mount Gambier.

The study revealed that the nitrogen loading of effluent applied was small in relation to the mineralisation which occurred when the pasture soil was ploughed and tilled. During the irrigation season there was negligible leaching of nitrate as the amount of water applied was in balance with evapo-transpiration losses. Before planting and after harvesting, however, substantial leaching of nitrogen occurred when the soil was bare, and rain could percolate freely through the sandy profile, leaching the considerable amount of nitrate which had accumulated as a result of mineralisation of soil organic nitrogen accelerated by soil disturbance.

The flux-weighted mean nitrate concentration in leachate at 1 metre depth was 69 mg N/L and the loading to the aquifer was 229 kg N/ha in 13.5 months as measured in monolith lysimeters. Nitrate concentrations in groundwater as measured in 18 piezometers in the 42 ha irrigation area averaged more than 25 mg N/L compared with three piezometers outside the perimeter of the centre-pivot irrigation area, which were less than 10 mg N/L.

Observations are continuing at this site, and the rate of decline of leachate and groundwater concentrations will be observed through this rotation cycle. Data collected at this site is being used by two groups of modellers to test their models of the fate of water and nitrogen in effluent irrigation; to compare three different techniques for monitoring of leachate; and to evaluate spatial variability of soil water and groundwater nitrate concentrations at paddock-scale. These results are to be published early in 1998. An interim report has been prepared by Dillon *et al.* (1997).

Pine plantations

Smethurst and Nambiar (1990) found that significant leaching (up to 150 kg/ha within three years) occurs only in the 4–5 years from the time of clear-felling of a pine forest to the closure of the canopy of the subsequent plantation. Enhanced mineralisation, lack of uptake of nitrogen in bare soil or by small saplings, and sandy free-draining soils on which pines are planted ensure that leaching of nitrate occurs during this period. Following canopy closure, pines are very efficient at taking up nitrogen which mineralises, and recharge rates are negligible (Holmes and Colville, 1970). Therefore over a 35 year clear-felling cycle, the average annual loading rate to the aquifer is estimated to be 4 kg N/ha.

There is evidence to suggest, on the basis of pine growth rates, that they have access to another water supply besides rainfall. If this is groundwater, pines could scavenge nitrate if it is present in groundwater. This also has major implications for water allocation policies in the region where demand is now thought to be close to the level of sustainable yield in some areas.

Leaching from other land uses

The nitrogen load from dryland grazing which occupies 63% of the study area, is assumed to be no greater than that of the dryland dairy pasture monitored at OB Flat. As the OB Flat site is in the highest rainfall area, it is expected that pasture production, therefore equivalent stocking rates for beef and sheep grazing are lower, hence livestock urine loadings are lower, and consequently leaching of nitrate may be less. A preliminary study undertaken on a dryland pasture 10 km north of Mount Gambier, found that lateral flow of water along a subsurface clay layer overlying an undulating surface of the Gambier limestone resulted in ponding in the lower parts of the paddock for intervals over winter. Although not proven, this has the potential for denitrification to occur, and it is possible that where organic rich topsoils become intermittently saturated that nitrate concentrations in groundwater could be substantially reduced.

Market gardening and field crops both have the potential for substantial mineralisation to occur following tillage. This is also true for viticulture at the time of establishment of vines. The management employed for these land uses is expected to dominate the amount of nitrogen leached to underlying aquifers. For example the time of tillage, duration of time that the ground is bare, the type and rooting depth of the crop, and the amount and timing of fertiliser applications will have a large influence.

There have been no field studies of the leaching of nitrogen from such land uses in the south east of South Australia, so Schmidt *et al.* (1997) were obliged to use data from literature, aware that these may be misleading, but are the best available estimates of nitrogen loading rates. Further studies in sensitive areas may be warranted to compare the effects of alternative management practices on crop production and groundwater protection. While market gardening occupies a very small fraction of land, it immediately flanks the Blue Lake, Mount Gambier's source of drinking water. Therefore it should rank as a relatively high priority for evaluation of impacts on groundwater quality.

Urban areas, prior to sewerage are expected to have gross loadings of 160 kg N/ha/yr based on typical per-capita loading rates (8 kg N/yr, after Dillon, 1997 (UNESCO report)) and population densities (20/ha). Therefore for example, the sewage from Mount Gambier (approx. 160 tonnes N/yr) now discharges to sea instead of to land.

Aquifer processes affecting nitrate concentrations

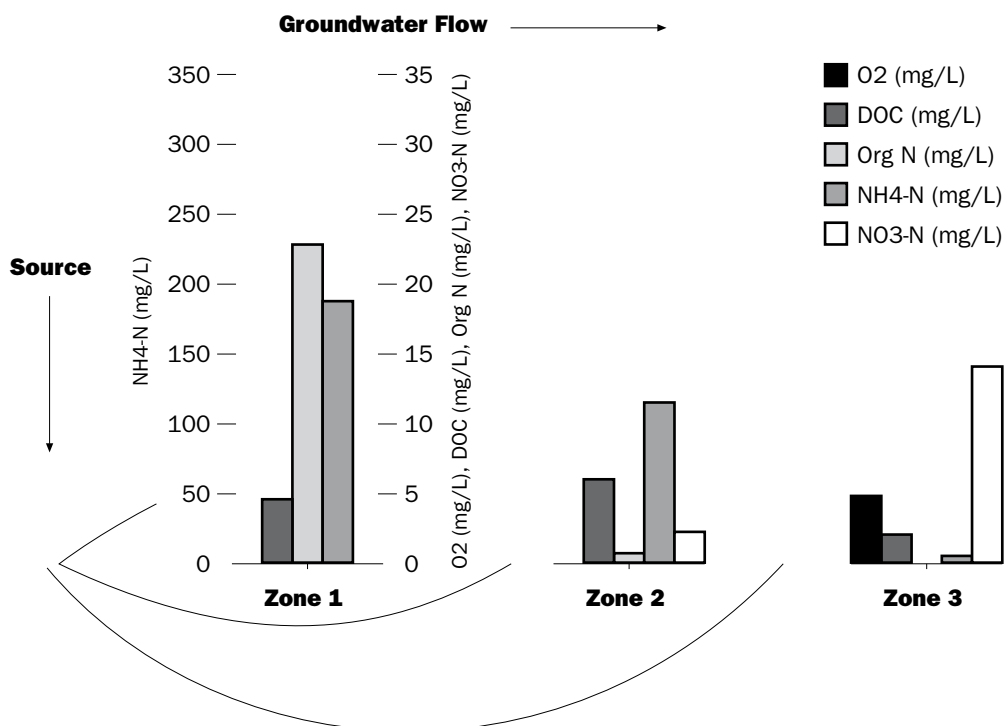
The Gambier limestone aquifer is unconfined and the water is aerobic except in the vicinity of significant sources of organic pollutants. A study by Richardson (1991) showed that cheese factory and abattoir effluent which entered the aquifer at Yahl over a period of 70 years, had created an anaerobic zone within the aquifer. In the core of this zone most of the nitrogen was still present in organic form. This was surrounded by an oxidation front where the dominant form of nitrogen was ammonium. In this zone the labile organic carbon present in the waste was depleted. Beyond this zone the groundwater became aerobic, the ammonium nitrified, and the nitrate concentration became stable as denitrification was inhibited by the presence of oxygen. Therefore the nitrate generated in the aquifer from decomposition of organic rich wastes is resistant to degradation and could be regarded as a conservative solute, capable of travelling within the aquifer at the speed of groundwater flow (Figure 9).

Where the watertable is very shallow there is some evidence of denitrification. Schmidt *et al.* (1997) show that the highest proportion of bores containing less than 2 mg N/L nitrate are in fact the shallowest bores (<10m). Dillon (1988) also showed this clearly for bore depth and depth to watertable for a sub-area containing several swamps. Groundwater in the vicinity of the coastal swamps and Dismal Swamp, which have the same land uses as adjacent areas (dairying and grazing), have significantly lower nitrate concentrations. Over winter when watertables rise into the top soil which is more organically-rich than the underlying soils, the surficial groundwater can become anaerobic, either in microniches or over expanses, allowing for microbial denitrification. Nitrate behaves as a conservative solute in all other areas of the Gambier limestone where the groundwater is aerobic. Nitrate is not an issue in the anaerobic waters of the Dilwyn Formation, and in places where seepage from the Gambier limestone to the Dilwyn occurs, nitrate will not be conserved.

Trends in nitrate concentrations

As described above, except in close proximity to point sources of contamination, nitrate concentrations are strongly auto-correlated. Even over ten years a regional survey was unable to detect a discernible trend in concentrations. For diffuse source contamination, investment in monitoring of trends on broad scale networks is unlikely to find uniform trends. However concentration changes from point sources may be detected with small spatial scales on a shorter time scale. A model was devised to assess the likely future of nitrate concentrations in groundwater.

Figure 9
Zonal chemical structure of a nitrogen plume at Yahl



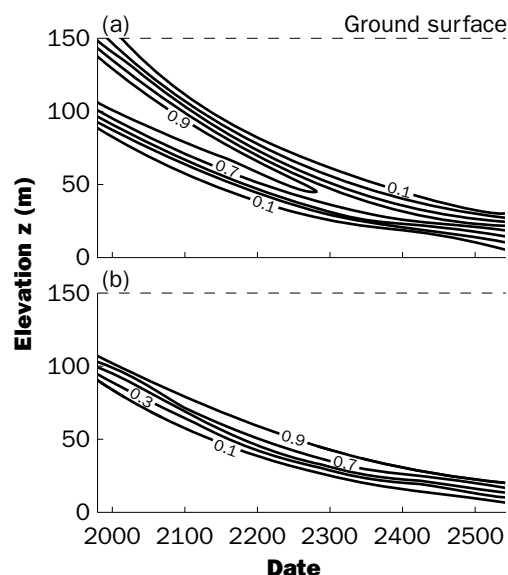
A numerical model DIVAST (**d**iffuse source **v**ertical slice **a**nalytical **s**olute **t**ransport) model was designed to assess the long-term regional changes in groundwater concentrations as a result of diffuse sources of nitrate (Dillon, 1988, 1989, 1990). It could also be applied to different management scenarios, such as how long will it take for a change in land management which reduces nitrate concentrations in recharge, to have an impact on groundwater quality. For reasons described in Dillon (1990) an analytical model was used, and applied to a typical cross-section along a flow path in the Gambier Limestone aquifer. The modelling predicts the potential low nitrate concentrations to increase the levels above the drinking water guidelines.

A key determinant of future nitrate concentrations is the amount of mixing which takes place in the vertical direction through the aquifer. Two different conclusions could be reached depending on assumptions on vertical mixing. If the aquifer is assumed to be stratified, then continued diffuse source pollution will increase the depth of the contaminated zone. Bores which are deeper in the aquifer will become contaminated, bores drawing a mixture of water from a depth interval of the aquifer will have increasing concentrations, and shallow bores will have stable concentrations similar to current values. If the aquifer is well-mixed vertically (which is less well supported by the nitrate concentration data) nitrate concentrations throughout the aquifer will continuously increase.

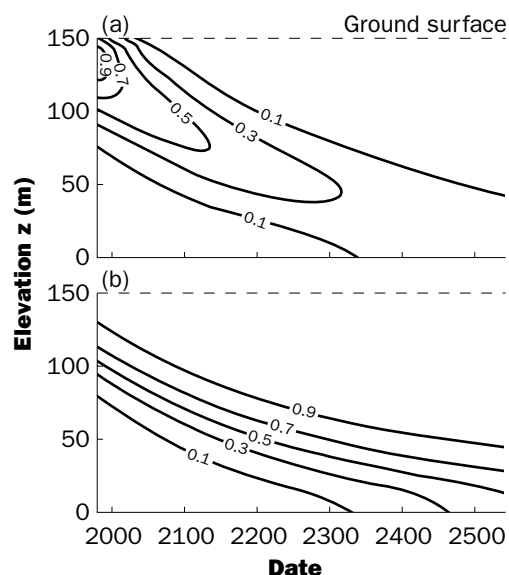
Two scenarios were evaluated for each assumption, one in which diffuse source contamination continued at current levels, and one in which, miraculously, the nitrate concentration in leachate suddenly became zero, and remained at that level thereafter. These scenarios represented worst case and best case management, to determine the influence of management on short and long-term concentrations of nitrate in groundwater. Concentration profile histories were projected forwards up to 500 years hence. The outcomes are shown in Figure 10 where concentrations in groundwater are scaled to the concentration of nitrate in recharge.

Vertical resolution of nitrate concentrations in groundwater, as provided by multi-port piezometers at a number of locations may give some clues as to the mixing coefficient to apply. Dillon (1988) provides a sensitivity analysis.

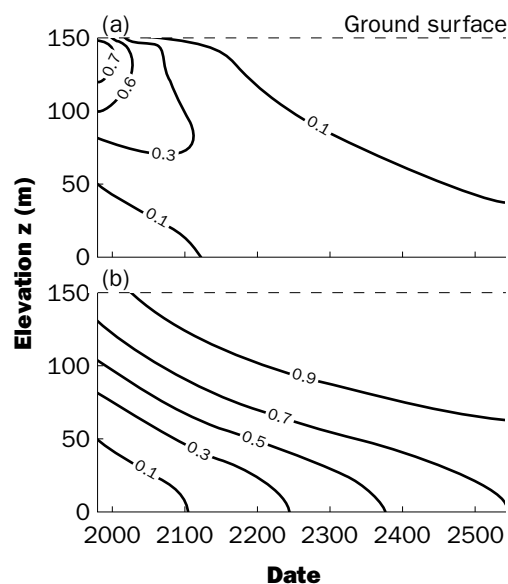
Figure 10
Projected nitrate concentration profile histories



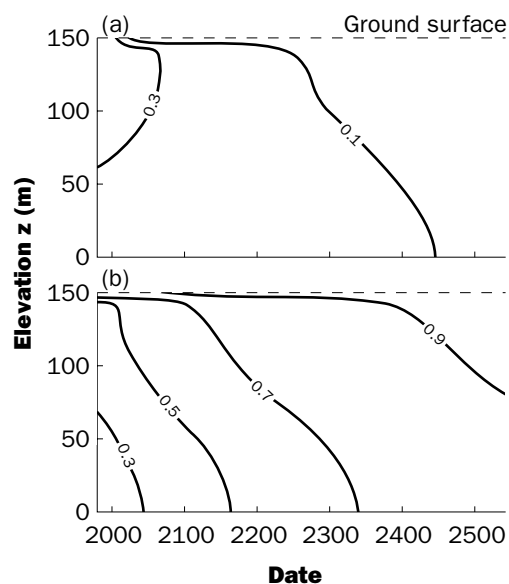
Predicted concentration profile history for 500 years with coefficient of vertical dispersion = $0 \text{ m}^2 \text{ yr}^{-1}$ unit recharge concentration (a) eliminated in 1986 (b) sustained.



Predicted concentration profile history for 500 years with coefficient of vertical dispersion = $1 \text{ m}^2 \text{ yr}^{-1}$ unit recharge concentration (a) eliminated in 1986 (b) sustained.



Predicted concentration profile history for 500 years with coefficient of vertical dispersion = $4 \text{ m}^2 \text{ yr}^{-1}$ unit recharge concentration (a) eliminated in 1986 (b) sustained.



Predicted concentration profile history for 500 years with coefficient of vertical dispersion = $16 \text{ m}^2 \text{ yr}^{-1}$ unit recharge concentration (a) eliminated in 1986 (b) sustained.

Source: Dillon (1988)

Land uses and aquifer nitrogen loading

Table 8 summarises the land uses and their nitrogen loads and is modified from Schmidt *et al.* (1997).

Table 8
Land use and estimated nitrogen loads to study aquifer area

Land use	Area (ha)	% of Area	N leaching rate * (kg N/ha/yr)	Annual N load (Tonnes N)	% of total N load
dryland grazing	386,000	66	7	2,700	37
forest	117,000	20	4	470	6
field cropping	44,000	7	60	2,640	36
dairy-dryland	16,000	3	**11	180	
	2.5				
dairy-irrigated	7,000	1	**25	180	2.5
potato/grazing	13,000	2	*50	650	8.8
viticulture	3,000	0.5	80	240	3.3
market gardens	1,500	0.3	130	200	2.7
urban	1,500	0.3	60	90	1.2
Total	589,000	100	(mean = 12.5)	7,350	100

* all loading rates based on estimates from literature except dairy-irrigated and dairy-dryland which have been measured at OB Flat (Pakrou, 1997), and preliminary measurements from potato/grazing rotation at Mingbool prior to completion of a rotation (Dillon *et al.*, 1997).

Source: Modified from Schmidt *et al.* (1997)

Management options

This section contains recommendations by Schmidt *et al.* (1997) to reduce the leaching of nitrate under various land management practices.

The overall management objective is:

“To reduce the concentration of leachate under any one land-use to below 10 mg/L nitrate (N) on average”

The fundamental strategy to minimise the leaching of nitrate is to ensure the generation or application of nitrogen in excess of plant requirements is minimised, and to ensure that there is adequate plant cover to take up the available nitrogen, thus minimising losses due to leaching. This will require detailed understanding of the nitrogen cycle processes involved in the different land management practices so that accurate nitrogen budgeting can be undertaken at the equivalent of a paddock scale.

Ultimately best management practices will have to be developed for each land management type, and these will have to be taken up by land managers, either on a voluntary basis or through mandatory regulation. Some recognised good practices to minimise nitrate leaching are outlined in the following sections.

General best practice methods

- Minimise fallow periods. If possible do not leave soil bare, especially after harvest. Plant a catch crop to take up the excess mineral nitrogen generated at this time.
- Direct drill crops and pastures wherever possible.
- Establish deep-rooted, winter active perennial pastures.
- Take care not to apply nitrogenous fertilisers in excess of the plant's ability to utilise the nitrogen. Apply fertiliser as close as possible to when the crop or pasture needs it most and when it is growing the fastest. Make sure that nitrogenous fertiliser is balanced with other essential nutrients and trace elements so that crops or pastures are able to make the best use of the nitrogen.
- Credit other sources of nitrogen in the soil from previous crops, particularly from legumes.
- Avoid disturbing the soil as much as possible, especially in early Autumn, to reduce the mineralisation of soil organic nitrogen just before the break of the season.
- Do not feed stock with nitrogen in excess of their needs for meat and milk production, as the excess nitrogen is excreted in urine and faeces, increasing the potential for leaching of nitrate.
- Irrigation should be scheduled to avoid keeping the soil unnecessarily close to field capacity, to minimise the potential for leaching.

All pasture systems

- Strip grazing for more even grazing and urine distribution patterns and to discourage stock camps.
- Careful management of stock density.
- Increasing conversion efficiency of herbage nitrogen into animal products by careful management of pasture composition, particularly clover composition, and controlling animal intake.
- Avoid nitrogen fertiliser application before the break of the season. Apply small amounts of fertiliser later when the pasture is growing vigorously.

Irrigated dairy pasture systems

- Irrigate while clover/ryegrass pastures are vulnerable to leaching losses over winter, avoid heavy grazing at this time.
- Grazing should be restricted to the driest ground and small applications of water should follow several days later.

Field cropping and market gardening

- Rates and timing of application of nitrogenous fertilisers and manure should be carefully matched to the crop requirements.
- Residual nitrogen from previous crops should be taken into account as part of the nutrient budget for crops.
- Leaving soil cultivation to the last possible moment before sowing.
- Practising minimum tillage, to avoid stimulating the soil microfauna to break down organic nitrogen in the soil to the soluble mineralised forms of nitrogen.
- Growing a catch crop after harvest or planting pasture simultaneously with a cereal crop.

Research and investigation needs

The primary need is to demonstrate the benefits of improved management practices in land use activities which have the greatest impact on the quality of groundwater. Best bet options, from the strategies suggested above should be trialed and compared with existing practices in replicated and scientifically controlled trials, in order to show the effects of these changes, and determine their impact on farm productivity as well as on groundwater quality. It is essential that primary producers are involved in these trials so that they can participate in experimental design and analysis of outcomes. This will help with implementation of methods which are found and agreed to be effective in protecting groundwater quality, and have acceptable consequences for productivity and effort in land management. Undertaking these trials will require new techniques for measurement of nitrogen fluxes to the watertable, particularly in the case of viticulture, where construction of economic monolith lysimeters of a suitable size without major soil disturbance will create non-trivial but technical difficulties. Innovative solutions will be required.

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Effluent disposal— Western Treatment Plant, Werribee, Victoria

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Introduction

The Western Treatment Plant is located 35 kilometres south-west of Melbourne and covers an area of approximately 10,850 ha. Approximately 50% of Melbourne's wastewater (500 ML/day), is treated at the WTP by the three major processes outlined below (Melbourne Water, 1995):

- **Lagoons:** An anaerobic and aerobic lagoon system covers approximately 1,667 ha and operates all year round, treating 70% of the total wastewater volume;
- **Land Filtration:** Land filtration operates during warmer months in spring and summer. The land filtration comprises 3,633 ha of 400 m bays that are flood irrigated with wastewater. This method of treatment accounts for 10% of the total annual WTP load; and
- **Grass Filtration:** Operates during the cooler months of autumn and winter and comprises 1,143 ha of 400 m bays that are flood irrigated. The grass filtration system accounts for treatment of 21% of the annual wastewater load at the WTP.

Groundwater water mound occurs beneath the sewage farm with the watertable typically at a depth of 2 m. During irrigation periods the watertable rises to within 0.3 m–0.6 m of the surface (Leonard, 1992).

Case study area—Delta trial site

Experimental studies on the fate of nitrogen during flood irrigation with raw effluent were carried out at the Delta trial site. This case study was selected to attempt to examine some of the processes which occur under effluent disposal conditions. This site comprises nine laser-graded bays with dimensions 400 m long and 40 m wide. The bays are graded at approximately one in 400. Five of these bays are within a forested site planted with a variety of eucalypts and pines. The remaining four bays occupy pasture which are flood irrigated between November and April. The pastures are used to support beef cattle and sheep that are grown and maintained at the Western Treatment Plant.

An earthen bank of approximately 0.5 m height is constructed between each bay to prevent inter-bay flow of effluent. An irrigation channel (140 South Carrier) is located at the northern end of the Delta trial site bays and is used to supply effluent to the bays during flood irrigation. The 140 South Carrier is supplied by the Western Trunk Sewer. Figure 11 shows the layout of the Western Treatment Plant and the Delta trial site and Figure 12 shows the layout of the nutrient balance study site.

Figure 11
Locality of the Western Treatment Plant study site

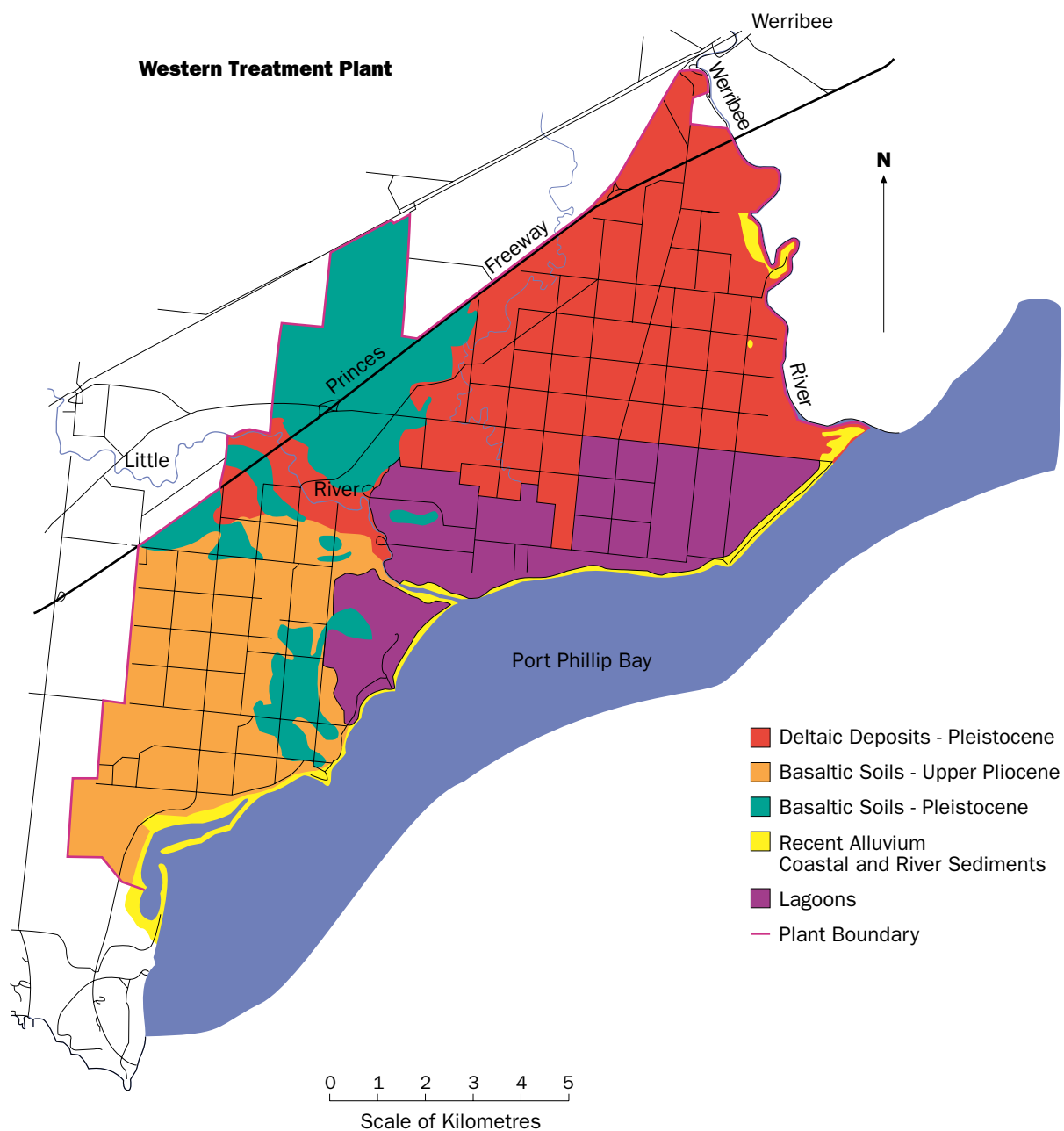
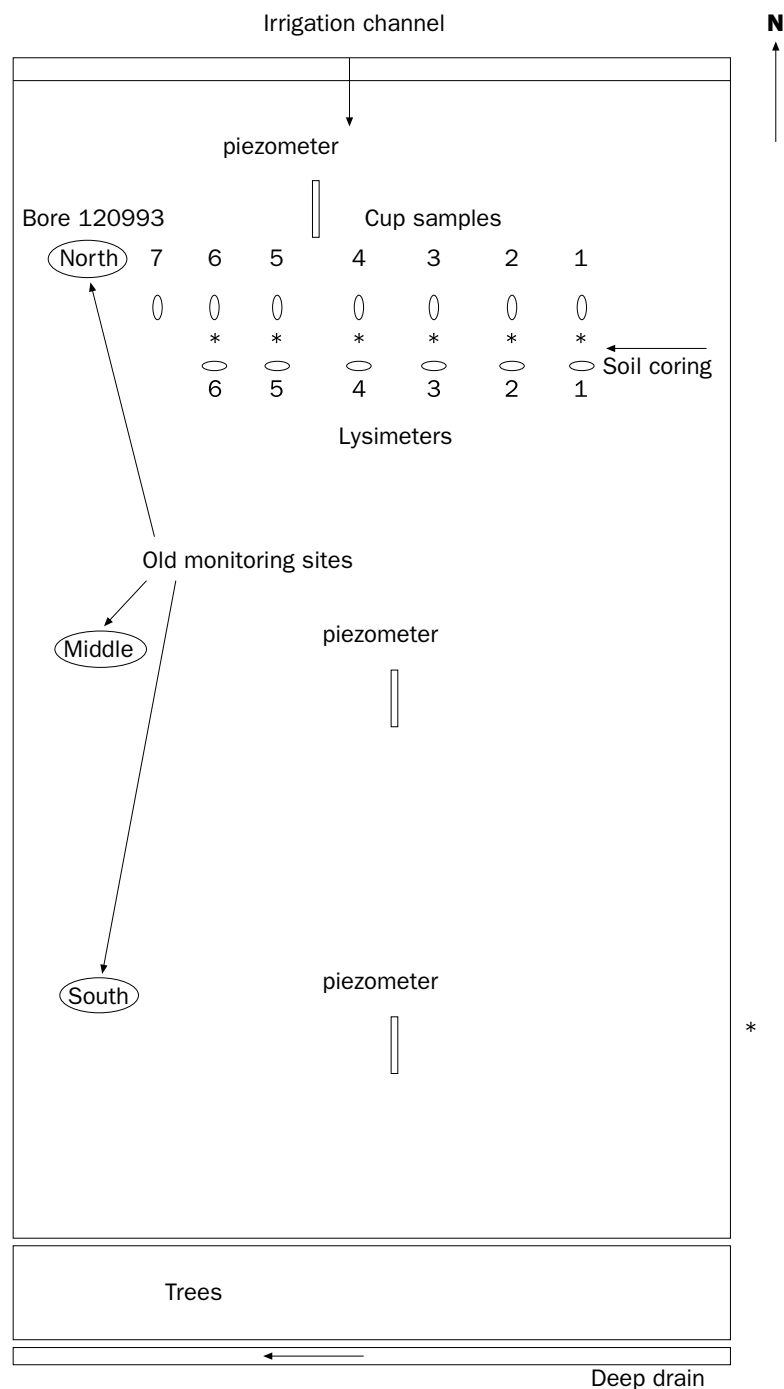


Figure 12
Layout of the Delta trial site



Land use history

The Delta trial site has been part of the flood irrigation, pasture grazing, program at the Western Treatment plant for 80 years. Each summer from November through to March, the bay has been irrigated with raw effluent, generally on a fortnightly basis. The average depth of effluent for a flood irrigation event is 100 mm of raw effluent and the irrigation volume is approximately 1 ML.

Irrigation events are not carried out on days following heavy rainfall. In 1989 the Delta trial site was laser graded to increase the uniformity of flow of irrigation water down the bay. There had been no laser grading prior to 1989.

Between irrigation events, cattle and sheep are allowed to graze on the irrigated bays. Animals are allowed access to the site three days after irrigation and are removed from the site the morning of an irrigation event.

Geological and hydrogeological setting

Climate

The climate on the Port Phillip Lagoon is temperate with warm summers and cool winters with maximum rainfall generally during winter. The topography has an influence on weather patterns, temperature variations and wind conditions.

A marked rainshadow on the leeward side of the Otway Ranges results in annual rainfall of less than 500 mm to the north of Werribee. The Werribee River sub-catchment receives approximately 990,000 ML per year of rainfall (Leonard, 1992). At the Western Treatment Plant, the average annual rainfall is approximately 460 mm.

Temperatures recorded at the Bureau of Meteorology, Werribee Weather Station show a mean annual of 8.9°C in July to a mean annual of 19.2°C in February with a mean annual average of 14.0°C.

Mean annual potential evapotranspiration is approximately 760 mm.

Geology

The Geology of the Werribee district including the Western Treatment Plant is covered by the Melbourne 1:63,360 Geological Map (SJ 55-1). There are four geological units that outcrop at the Western Treatment Plant. These are:

- Newer Volcanics of Upper Pliocene to Lower Pleistocene age (1.7 to 1.9 Million years ago);
- Deltaic Deposits—Pleistocene age (0.1 to 1.8 Million years ago);
- Recent alluvium (river deposits)—Recent age (0 to 0.01 Million years ago); and
- Recent coastal swamp, beach sand deposits—Recent age (0 to 0.01 Million years ago).

Basement in the vicinity of the Western Treatment Plant is undifferentiated Ordovician–Silurian sediments. Devonian granite (You Yangs) and acid intrusive (South Yarra) basement wedge-out near the western boundary of the Western Treatment Plant.

The Newer Volcanics are overlain by the Deltaic deposits in the vicinity of the Western Treatment Plant. They are on average approximately 60 metres thick at the WTP.

The Deltaic deposits of the Werribee River form a fan shape and comprise brown to grey-brown silt, gravels, sand, silty sand lower in the sequence with minor gravel and sand levees and abundant carbonate nodules. The top of the fan is situated north of the Princes Highway. At the coastline the deposits extend for approximately 15 kilometres.

Alluvium deposits comprising brown to dark grey silt, sandy silt with minor sand and gravel are confined to river terraces, particularly in the upper reaches of the Werribee River delta. Coastal swamp and beach sand deposits are confined to the middle to lower reaches of the Werribee delta and comprise fine sand and silt often with shell beds to well sorted calcareous quartz sand.

Hydrogeology

The Werribee Delta is a coastal aquifer, located on the deltaic deposits of the Werribee River mouth and covers approximately 117 square kilometres. The Werribee aquifer is unconfined to semi-confined with a thickness that varies up to 40 metres in the area along the coast immediately east of the present Werribee river course. In general, the depth to the aquifer is approximately 7 metres, but below the Western Treatment Plant it is considerably smaller, typically less than 2 metres resulting in a groundwater mound (Leonard, 1992).

The characteristics of the Werribee Delta aquifer are variable, reflecting the nature of the lithology. Hydraulic conductivities in the coarser horizons range from 10 to 15 m/day, while an average hydraulic conductivity of 5 m/day can be assigned elsewhere. Bore yields in the Werribee Delta sediments can be as high as 15 L/sec though a representative yield would be less than 5 L/sec. Specific yields of the coarser units would be approximately 15 to 20% with a bulk average estimate less than 10% ascribed elsewhere (Leonard, 1992).

Groundwater salinity in the Werribee Delta sediment aquifer varies from 500 to 6,000 mg/L total dissolved solids (TDS) with sodium chloride (NaCl) being the major salt type. Lower salinities are typical of the coarser, channel sediments. Nitrate concentrations range from below detection limit to as high as 39 mg/L. The pH of the groundwater varies from 7.8 to 9.5 with average value of 8.3 (Leonard, 1992).

Recharge to the Werribee Delta aquifer occurs due to:

- surface infiltration; and
- interflow from the underlying Newer Volcanic aquifer as evidenced by the hydraulic gradients between the Werribee Delta aquifer and the Newer; this recharge component has not been quantified to date.

Leonard (1992) estimated direct annual groundwater recharge rates to the Werribee Delta aquifer for 3, 5 and 10% of the annual rainfall to be approximately 1,800, 3,000 and 6,000 ML/day respectively.

Discharge from the Werribee Delta aquifer occurs into the lower reaches of the Little and Werribee Rivers and along the coast as well as near offshore into Port Phillip Bay and an estimated sustainable yield for the Werribee Delta aquifer was determined by Leonard (1992) to be 3,000 ML/year.

Soils

Three major soil type are recognised throughout the Western Treatment Plant (Melbourne Water, 1994). They are:

- Delta soils;
- Basalt Plain soils; and
- Tongue Flow soils.

The delta soils are the most significant soils for land treatment processes. Soil permeability varies from 25 to 30 mm/day. Soil texture varies from red-brown fine sandy loams to red and grey-brown fine sandy clay loams. A sharp boundary delineates the change to a red-brown medium clay from the variable textured soils. Underlying these soils are variable textures but generally reddish-brown fine sandy clay loam or heavy clay loam which changes to a fine sandy clay loam. Heavy clays are present in the deeper soils (Melbourne Water, 1994).

The Basalt Plain soils have low permeabilities, generally between 1 and 3 mm/day. Red-brown light clays are typical of the surface soils, gradually hanging to a yellow brown medium clay underlain by heavy clays (Melbourne Water, 1994).

The Tongue flows have similar permeabilities to that of the Basalt Plain soils and are associated with undulating terrain and rocky outcrops. A red-brown clay loam is typical of the A horizon which changes sharply to a red-brown medium clay underlain by relatively fresh calcareous clay and basaltic rock (Melbourne Water, 1994).

The test site at the Delta trial site is situated within the Delta soils.

Estimation of historical nitrogen loading at Delta Trial Site

Analyses of the nitrogen concentration of raw effluent prior to flood irrigation indicates that the effluent contains nitrogen concentrations shown in Table 9. These results were gained from analyses of effluent prior to 10 irrigation events over the 1995–96 and 1996–97 irrigation season.

Table 9
Raw effluent nitrogen concentrations

Nitrogen Form (as N)	Concentration (mg/L)
Total Nitrogen	41
Organic-Nitrogen	10
Total Kjeldahl Nitrogen	41
Ammonium Nitrogen	31
Nitrate Nitrogen	0.1

Using the average concentration of nitrogen from Table 9, the total nitrogen loading during an irrigation event is approximately 41 kg. The area of the trial site is 1.6 ha. This gives an average nitrogen loading of 2.56 kg(N)/ha per event. If in an irrigation season, the plot is irrigated 10 times this gives a annual nitrogen loading of 256 kg(N)/ha/yr. This loading does not include additional nitrogen from animal manure during grazing periods at the site. Over 80 years of flood irrigation the amount of nitrogen added to the site is in the order of 20,480 kg.

Groundwater contamination processes

The results of an intensive groundwater monitoring program carried out at the Delta trial site over the 1995–1996 irrigation season (DNRE, 1997a), showed that significant connection occurs between the groundwater and surface water at the site. The connection is considered due to the presence of soil cracks resulting in macropore. A second groundwater study at the site over the 1996–97 irrigation season (DNRE, 1997b) confirmed that significant sub-surface lateral flow occurred during irrigation events. This almost immediate response of groundwater to flood irrigation at the site represents a significant pathway for the contamination of the groundwater with nutrients in the effluent.

There are several hypothetical nitrogen processes that could occur during flood irrigation at the Delta trial site, some of which have the potential to lead to nitrate contamination of groundwater:

- nitrification: ammonium in the raw effluent could be oxidised to nitrate by soil microorganisms;
- leaching of ammonium: the effluent could move quickly in soil cracks and via macropore flow result in ammonium being detected in groundwater;
- volatilisation of gaseous ammonia: ammonium in raw effluent can be volatilised and lost to the atmosphere; and

- denitrification: denitrification may be considered a significant pathway for the loss of nitrate in the case study as the soil has received a constant source of organic carbon, which may be in sufficient concentration to allow the reduction of nitrate within the soil profile or in the upper portion of the aquifer.

Nitrogen balance at Delta trial site

Table 10 shows the nitrogen fate (as % of applied ^{15}N) from three flood irrigation events using mini-lysimeters at the Delta trial site.

Table 10
Nitrogen balance at Delta trial site

Nitrogen	% of Applied ^{15}N
Leaching Loss	2
Soil Recovery	35
Plant Recovery	19
Accounted for	56
Unaccounted-for	44

The unaccounted-for nitrogen could be attributed to volatilisation of ammonia gas or gaseous loss due to denitrification. The major points of the nitrogen balance are as follows:

- Of the 2% of nitrogen that was recovered in the drainage water of the lysimeters, nitrate was the major form measured. Nitrate concentrations ranged from 8.0 to 46.5 mg/L;
- Of the 35% of applied ^{15}N that was recovered from the soil after destruction of the lysimeters, in each lysimeter more than 90% of the radiolabelled N recovered was in the form of soil organic nitrogen; and
- Of the 44% of radiolabelled nitrogen that was not accounted for, this could be due to volatilisation of ammonium and/or gaseous losses due to denitrification. The role of denitrification was not able to be addressed during this experiment.

Nitrate contamination on groundwater

There are limited available groundwater monitoring data available for the Western Treatment Plant at Werribee, despite the long-term effluent disposal at the site. Management of the site has precluded the establishment of long-term Government observation bores at the site. However two bores on the immediate down gradient side of the site have no nitrate recorded.

Anderson (1988) indicated that ammonia is the dominant form of nitrogen in the groundwater in the deltaic sediments with concentrations up to 40 mg/L, while nitrate and nitrite occur only in the underlying basalt. NO_3 in the basalts is recorded at concentrations between 0 and 30 mg/L.

It is interesting to note from the Government bore data base that NO_3 concentrations in groundwater beneath agricultural areas on the northern side of the Werribee River, outside the effluent disposal area, are between 9 and 23 mg/L NO_3 . This is considered more likely to be due to agricultural processes and fertiliser applications than from the effects of the effluent disposal.

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Septic tank study— Nepean Peninsula, Victoria

Introduction

The Nepean Peninsula is located to the south of Melbourne separating Port Phillip Bay from Bass Strait.

An investigation of groundwater quality by Sinclair Knight Merz on behalf of South-East Water was undertaken in 1996 to determine the impacts on groundwater quality from septic tank seepage on the Nepean Peninsula (SE Water, 1996).

Land use history

There is a high demand for groundwater in the Nepean Peninsula, generally in the less-urbanised areas. Over 800 groundwater bores are registered for stock and domestic use. The coastal fringe of the Peninsula is densely populated with large increases in population over peak holiday and summer periods. The central area of the Peninsula, known as ‘The Cups’, is used mainly for grazing, market gardens and recreation.

Reticulated potable water is supplied to the Peninsula area, but the area is not sewered and wastewater is treated generally by septic tank systems.

Geological and hydrogeological setting

The Tertiary sediments to the east are divided into a number of distinct units, the thickest being the Fyansford Formation, consisting of a calcareous clayey silt which forms the base of the sedimentary sequence and is up to 400 m thick. This is conformably overlain by sediments of the Brighton Group which are composed of up to 45 m of sands and clays. This, in turn, is conformably overlain by the Pliocene Wannaeue Formation and then the Pleistocene Bridgewater Formation. The Wannaeue Formation consists of sandy calcarenite, shelly sands, muds and clay. The Bridgewater Formation consists of thin beds of fine to medium grained moderately cemented quartz-carbonate sand and aeolianite showing bedded associated with dune systems. The aeolianites are reported to have up to five paleosols which could alter groundwater flow and potentially develop perched watertables. This formation can reach a thickness of 84 m. Thin minor Recent deposits occur along the coastal margins.

The upper sandy horizons of the Tertiary sediments consisting of the Brighton Group, the Wannaeue Formation and the Bridgewater Formation, are in hydraulic connection over most of the area and are considered to act as a single aquifer system. The underlying Fyansford Formations acts as a regional aquitard.

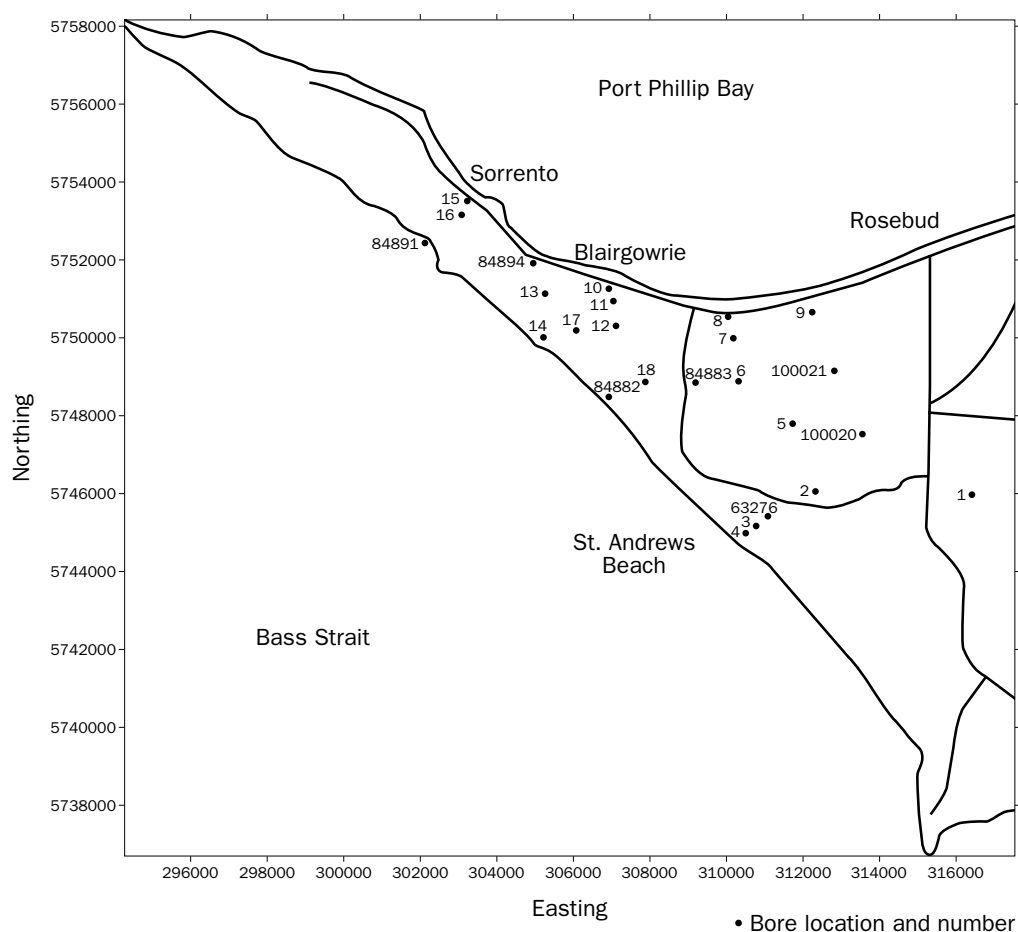
Fresh water recharge of the aquifers occurs predominantly to the west of the Selwyn Fault, in ‘The Cups’ area. Groundwater flows radially from the recharge area and the sandy nature of the superficial soils across the Peninsula results in additional recharge of the aquifer from infiltration of precipitation, stormwater and septic tank effluent. Additional recharge mechanisms become increasingly important towards the west as the influence of ‘The Cups’ recharge zone wanes.

The fresh water from the recharge areas sits as a lens above saline groundwater which has salinities similar to that of the sea. The area can be compared to an island recharge environment. The Ghyben Herzberg relation (Todd, 1980) estimates that for every metre head of freshwater above sea-level, there is approximately 40 metres of freshwater below sea-level, the boundary between the two layers being transitional.

The potentiometric head in the shallow aquifer is generally at least 2 m above sea level and therefore there is potentially a significant groundwater source in the Nepean Peninsula.

A groundwater monitoring bore network was constructed throughout the Peninsula from Rosebud to Sorrento. Eighteen groundwater observation bores were constructed and supplemented by seven existing Government-owned observation bores. The location of observation bores is shown in Figure 13.

Figure 13
Location of Nepean Peninsula groundwater observation bores



Observation bores were sampled on a monthly basis for 12 months from June 1995 to May 1996. Groundwater samples were analysed for the chemical and microbiological parameters shown in Table 11.

Nitrate groundwater contamination

Groundwater analyses for nitrate and nitrite for the observation bores are shown in Table 12. Analyses show that there is substantial contamination of groundwater resources in the Peninsula from nitrate, although these concentrations vary both spatially and temporally.

Table 11
Chemical and microbiological groundwater analyses

Chemical Analyses		
pH	Electrical Conductivity (EC)	Total Alkalinity (as CaCO ₃)
Hardness (as CaCO ₃)	Chloride	Sulfate
Calcium	Magnesium	Sodium
Potassium	Iron (undigested)	Nitrate (as N)
Nitrite (as N)	Silica	Total Phosphorus
Total Dissolved Solids (TDS)		
Microbiological Analyses		
<i>Escherichia coli</i>	Faecal Streptococci	Total Coliforms

Table 12
Nitrite and nitrate (NO_x) concentrations in groundwater samples

Bore ID	NO _x – N (mg/L)			Bore ID	NO _x – N (mg/L)		
	Number of	Minimum	Maximum		Number of	Minimum	Maximum
	Positive Samples (total samples)				Positive Samples (total samples)		
Upgradient Rural Bores							
1	11 (12)	ND	76	100020	2 (11)	ND	47
2	2 (12)	ND	8.2	100021	2 (12)	ND	42
5	2 (12)	ND	69				
Urban Bores							
63276	2 (11)	ND	0.73	9	5 (11)	ND	56
84882	12 (12)	0.58	7.1	10	11 (11)	0.5	5.9
84883	11 (11)	1.7	60	11	11 (11)	0.34	2.8
84891	11 (11)	0.95	3.2	12	12 (12)	1.7	4.5
84894	1 (11)	ND	2.5	13	4 (12)	ND	1.1
3	11 (12)	ND	16	14	12 (12)	10	13
4	12 (12)	6.7	16	15	1 (11)	ND	3.8
6	3 (12)	ND	58	16	1 (11)	ND	2.7
7	12 (12)	2.5	58	17	11 (12)	ND	5.8
8	3 (11)	ND	47	18	11 (12)	ND	5.4
Reference Bore							
23	1 (4)	ND	1.3				

Notes: ND = concentration below detection limit of 0.05 mg/L

Nitrogen loads

Nitrogen concentrations were found to be above the detectable level (0.05 mg/L) in all of the coastal bores at various times during the study. All bores on the ocean side showed a consistent detection of nitrogen throughout the sampling period. The bores on the Bay side of the Peninsula demonstrated a more irregular detection pattern.

The peak levels of nitrogen seen in the north 1 and 2 flowlines during September have significantly effected the estimate of nitrogen entering the Bay. The nitrogen loads in these two flowlines for the one month of September account for almost 80% of the total estimated influx of nitrogen to the Bay from a groundwater source.

The total annual load of nitrogen from groundwater sources has been estimated at approximately 5.7 tonnes per year. Other sources of nitrogen input to the Bay could be from surface drains, which discharge directly to the Bay at various locations. These sources have not been included in the figures presented as the figures are strictly related to groundwater discharge to the Bay.

A similar nitrogen load is discharged to the ocean as to the Bay. The South 1, 2 and 3 flowlines account for the majority of the load, with the Blairgowrie and St. Andrew's Beach areas (South 1 and 3) contributing the most. The estimated total load of nitrogen reaching the ocean is approximately 5.1 tonnes per year.

It has been estimated that groundwater discharging from the margins of the Nepean Peninsula contributes approximately 10.8 tonnes of nitrogen to the adjacent marine environments each year.

Groundwater contamination processes

The sediments composing the upper levels of the Nepean Peninsula are highly conductive to groundwater fluxes. Estimates of horizontal hydraulic conductivity (or permeability) are in the order of 25 to 30 m/day (Leonard, 1992). In a sandy environment, the vertical hydraulic conductivity is likely to be approximately an order of magnitude less than the horizontal hydraulic conductivity. Vertical hydraulic conductivity is affected by depositional processes that sort particle sizes. Alternating graded particle bands results in a more uniform distribution horizontally than vertically, therefore, the hydraulic conductivities are higher in the horizontal direction compared to the vertical direction.

The vertical hydraulic conductivity is also related to the moisture content of the soil. Saturated soils have a higher hydraulic conductivity than unsaturated soils. This is due to the retention of water within the unsaturated soil matrix through particle and surface forces. In a saturated soil, these retentive forces are fully met. Additional water can, therefore, move relatively freely through the matrix. As the septic tank seepage lines will always produce a higher level of soil moisture in the underlying soils than in surrounding areas, when rainfall rates are in excess of evapo-transpiration rates, rapid leaching of contaminants from septic tank seepage lines to groundwater is likely to occur.

The amount of infiltrating water reaching the groundwater will depend upon the depth to the watertable and the moisture deficit of the soils in the unsaturated zone of the soil horizon.

As the soils of the Nepean Peninsula are almost entirely sandy and free-draining, there will be little surface run-off generated through saturation of the upper soil profile. Infiltration is, therefore, likely to be favoured during the months in which evapo-transpiration rates are typically lower and rainfall higher, namely winter and spring. Rainfall events of limited intensity during the summer period could be expected to be largely taken up by evapo-transpiration influences at the surface and not reach the groundwater system. Low moisture content in the unsaturated zone will also restrict the movement of a wetting front in this zone.

The lack of significant clays within the soil profile will restrict the capillary rise emanating from the watertable into the unsaturated soil zone, therefore, limit the removal of groundwater from the watertable through evaporation. Evaporation of the capillary water produces deposition of dissolved salts in the unsaturated soil horizon.

The capillary rise will be limited, rather than negated, so some potential remains for the accumulation of salts within the unsaturated soil horizon. Salts accumulating in the soil during the high evaporation months could be expected to be leached back to the watertable through percolating rainwater. This suggests that salts are likely to accumulate over the summer period and be leached back to the groundwater during significant rainfall events. The time series of TDS show a general rising trend in groundwater salinity in the months following December. These results may have been influenced by the rainfalls occurring just prior to the January sampling and during the February sampling runs. The increase in groundwater salinity generally corresponds to decreasing watertable levels and the rising salinity trend generally extended through to the April sampling episode, when groundwater salinity began to decline.

The rise in TDS levels over the summer period may also be influenced by the influx of septic tank effluent to the groundwater. The TDS concentration of the septic tank seepage may be greater than the background level in the groundwater in many instances (Brouwer and Bugeja, 1983). The reversal of the rising trend in groundwater TDS levels, marked in the April sampling results, may be due to a combination: of decreased septic tank seepage; increased throughflow from increased precipitation; and declining soil salinity due to a flushing effect.

The main peaks in NO_x concentrations occur in the eastern zone of the study area and are limited to the months of September and October. It is expected that these levels relate to fertiliser applications within the horticultural and farming areas. No confirmatory evidence is available to support this assumption, however, the peak levels occur for only a short period of time coinciding with a presumed period of spring fertilising. Levels of NO_x seen in these bores do not occur elsewhere in the study area for the remainder of the sampling program. The high levels detected in these bores are, therefore, thought to be due to a short term event localised to this area.

The velocity of groundwater carrying contaminants can be estimated from Darcy's equation of groundwater flow, which states that:

$$Q = K.i.A$$

where

Q = flow

i = the hydraulic gradient or change in head over distance, and

A = cross sectional area through which the flow occurs.

The Darcy velocity (of groundwater) is given by:

$$v = K.i$$

A measure of the actual velocity, according to the Darcy equation can be derived by dividing the Darcy velocity by the porosity ($V = v/n$). For the sandy sediments found in the Nepean Peninsula, this gives an estimated actual velocity in the order of 1 to 100 mm per day. This low figure is largely the result of low hydraulic gradients that exist within the Peninsula.

The detection of high levels of NO_x at the coast over relatively short periods of time is not supported by this calculated velocity. Tóth (1963, cited in Freeze & Cherry, 1979) has suggested that a number of flow systems may exist within the overall groundwater flow system, particularly where local relief is significant. Local flow systems overlie intermediate and regional flow systems. The detection of high NO_x levels in bores monitoring the north 1 and 2 flowlines to the east indicates either a more rapid transport mechanism than is suggested by the Darcy velocities or a broader application of fertilisers than expected. These high levels of NO_x are taken to suggest the existence of alternative flow mechanisms operating in the shallow groundwater system.

Groundwater analyses for September show elevated levels of NO_x in most bores in the north east part of the study area. Bores 100020 and 100021 are screened approximately 15 metres below the watertable. Bores drilled for this study screen the upper 5 metres of the groundwater. The deep screened bore immediately downgradient of the Boneo Road horticultural area shows a NO_x peak, while the other deep screened bore (Bore 100021) further downgradient does not. The shallow screened coastal bore (Bore 9) also shows a high NO_x level. All shallow bores in the adjacent northerly flowpath demonstrate high NO_x concentrations. This flowpath extends from 'The Cups' to the margin of the Bay.

Groundwater analyses for October show elevated NO_x concentrations in the three deep screened bores (Bores 100020, 100021 and 84883), but the shallow screened bores have returned to pre-September, baseline levels. This suggests that NO_x movement within the sediments may be orders of magnitude higher than the calculated Darcy velocities suggest. This result points to at least two, somewhat separate, contamination transfer paths from the source to the groundwater discharge points.

This may be achieved by a number of mechanisms including preferential or macropore flowpaths, which can accommodate much larger velocities than the general matrix. The shallow area of the watertable may also be affected by localised and transient elevations in the watertable induced by infiltration. The NO_x contaminant plume may then move as a pulsed flow. If only partial vertical mixing occurs, or a number of laminations of flow exist within the groundwater system, the deeper screened bores will intersect a different sample of groundwater to the shallow screened ones. This could account for the distribution in high NO_x levels found during the September and October sampling runs.

Although the high NO_x levels seen in the east of the study area are the most significant found, the source of the contamination is almost certainly of an origin other than septic tank seepage. The remaining bores in the study show various levels of NO_x , ranging from below detectable limits to approximately 20 mg/L. Most of the remaining bores show relatively stable NO_x levels, with small oscillations about a mean value. Minor rises in NO_x concentration are seen in many bores over the summer period, particularly during February. Many of these bores show a rising NO_x trend over the summer period.

Bore 3 in the St. Andrews Beach area, shows a marked rise in NO_x following the summer peak period. Levels remain elevated until the May sampling run. Bore 4, located nearer the coast, does not show this trend, however, NO_x levels are substantially higher than that seen in the background bores (Bores 2 and 63276). This indicates there is contamination of the groundwater in the St. Andrews Beach area from a NO_x source, presumed to be from septic tank seepage.

The Blairgowrie Ocean Beach area monitored by Bore 14 shows continual high NO_x levels around 12 mg/L. The NO_x levels in this bore are reasonably constant with a slight rise in February following a slight decline in January. This area is densely populated with homes and the surface soils are particularly sandy. The centrally located bores (Bores 12 and 17) show reasonable levels of NO_x , generally below 5 mg/L. Bore 17 shows much greater variation in NO_x levels with a peak concentration detected in September.

Flowpath South 2 incorporates Bores 84883,18 and 84882. In general, there is an increase in NO_x concentration along the flow path and a rising trend in the more coastal of the two bores over the summer period. This is presumed to be related to an increased occupation rate, therefore, increased septic tank loads.

The NO_x levels alone do not provide conclusive evidence of contamination of the shallow aquifer with seepage from septic tanks. For many of the bores, there appears to be a relationship between presumed peak population loads and elevations in NO_x levels. However, there are bores in which the expected trend is reversed. As many bores are located within the urban areas, localised events, such as garden watering and fertilising in the vicinity of the bores, may have a significant effect on the levels detected. Fertilising of urban green areas over sandy soils has been shown to contribute to the inorganic loads placed on a sandy aquifer in Western Australia (Sharma *et al.*, 1996).

Discussion

All bores within the study area showed detectable limits of NO_x at some time during the sampling period. In most groundwater samples, the detected levels in nearly all bores were below the ANZECC (1992) drinking water guidelines of 10 mg/L.

Summer contaminant loading from peak population levels coincides with high evapo-transpiration rates and low rainfall. This assists in the localisation of septic tank seepage within close proximity of the seepage lines as transpiration is favoured over leaching. Where the depth to watertable is greater, there is less likelihood that the seepage will reach the watertable, except when leached by substantial rainfalls. It is possible that groundwater quality with respect to NO_x may improve over the summer and decline again as infiltration increases leaching rates.

NO_x levels in a number of bores have peaks that are many times above the recommended level, although these levels do not persist. In these areas, groundwater is unsuitable for human consumption, based on nitrogen alone, at various times of the year.

The general lack of very high levels of indicator microorganisms suggests that either the process of contamination is of a consistent, low level nature or that the frequency of the sampling program was unable to detect short-lived, peak contaminant loads.

Groundwater is extensively used within the Nepean Peninsula. In the eastern horticultural area, groundwater is an important source of irrigation water. In the remainder of the Peninsula there are over 800 stock and/or domestic bores extracting groundwater. Many of these are probably utilised for garden watering, however, the level of usage of groundwater for human consumption is unknown. As a reticulated water supply system operates throughout the Peninsula, the use of groundwater for human consumption is considered to be low at present. This is due to the salinity of the groundwater being considerably higher than the reticulated supply. While the groundwater is potable, it is generally above 500 mg/L TDS and would taste relatively salty.

Where bores are placed in close proximity to septic tank seepage lines, a potential risk exists for extraction of contaminated groundwater. This can occur when subdivision of land into small blocks is undertaken with little planning for placement of septic tanks and groundwater extraction bores.

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Septic tank study— Venus Bay and Sandy Point, Victoria

Introduction

Venus Bay and Sandy Point are two coastal towns in south-eastern Victoria. Groundwater investigations have been carried out at Venus Bay since 1992, with a small network of four groundwater bores being installed. This monitoring network was extended in 1991 and 1992 to include 22 monitoring bores (Southern Rural Water, 1996).

Land use history

The lack of utilities such as reticulated water and sewerage have resulted in the conjunctive use of the Venus Bay and Sandy Point aquifer for septic waste disposal and water supply. The advent of increased development has seen the proliferation of both septic tanks and groundwater extraction bores.

Water for drinking and domestic use is generally obtained from rainfall collected in rainwater tanks. There is anecdotal evidence which suggests that a number of landholders supplement tank water supplies with groundwater, presenting a potential public health and safety problem (Southern Rural Water, 1997).

Geological and hydrogeological setting

Venus Bay and Sandy Point have similar hydrogeological settings. Both areas are located on calcareous and quartzite sand peninsulas. The high quality groundwater found at these locations is derived from infiltration of rainfall into the sandy sediments. The saltwater interface is held at bay through a combination of density effects and continual throughflow of freshwater to the sea.

The lower density of freshwater relative to seawater results in a freshwater lens sitting above and within the surrounding sea water. The vertical depth of the saltwater–freshwater interface depends on the elevation of the groundwater above sea level. The depth of the interface is approximately 40 times the elevation of the groundwater above sea level.

Groundwater quality investigation

The results of groundwater analyses for nitrate and nitrite for the period 1994 to 1996 are shown in Table 13.

Groundwater contamination processes

The hydrogeological setting at both Venus Bay and Sandy Point results in a delicately balanced groundwater system. Interference with the natural processes could precipitate adverse effects on groundwater quality.

The predominance of sands and the absence of substantial amounts of clays in the upper sections of the lithological profile results in little capacity for attenuation of the effluent seeping from the septic tanks. Leaching by infiltrating water rapidly transports contaminants from the septic tank environment to the shallow groundwater system.

Table 13***Nitrate/nitrite in groundwater samples from Venus Bay and Sandy Point (January 1994–April 1996)***

Bore ID	Number of Positive Samples (Total Number of Samples)	Nitrate + Nitrite – N (mg/L)	
		Minimum	Maximum
Venus Bay Bores			
94802	1 (3)	ND	0.27
94803	1 (3)	ND	0.074
94804	2 (3)	ND	1.5
94807	2 (3)	ND	1.2
94809	2 (3)	ND	0.5
94814	2 (3)	ND	0.78
94815	1 (3)	ND	0.84
114158	3 (3)	16	47
Sandy Point Bores			
100975	1 (3)	ND	0.096
100976	1 (3)	ND	0.045
100977	1 (3)	ND	0.005
100978	3 (3)	0.67	9.8
100979	3 (3)	1.0	11.0
100980	1 (3)	ND	0.017

Note:ND = concentration below detection limit of 0.05 mg/L

The low horizontal gradients that exist in the groundwater at these localities results in relatively slow movement of water through the aquifer, despite the sediments having a relatively high porosity. The slow movement of the groundwater provides time for natural attrition of the enteric bacteria within the aquifer. Persistent organisms, such as viruses, and chemical contamination may not degrade before entering the zone of influence of adjacent extraction bores.

Extraction of groundwater via pumping produces a localised cone of depression in the watertable. This results in the movement of water from surrounding areas toward the pumping bore by increasing the horizontal gradient. Increasing the extraction rate or the duration of pumping produces a more intense, local decline in groundwater heads and increases the zone of influence of the bore. Water will therefore be drawn from a wider area and the possibility of extracting contaminated water increases.

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Septic tank study— Benalla, Victoria

Introduction

Benalla is a town of approximately 9,200 people straddling the Broken River in Victoria (Figure 14). Several areas within the city are unsewered, and in 1993 a groundwater investigation was carried out on an area between the town centre and the Hume Freeway which contained approximately 40 septic tanks.

Hydrogeology

The area investigated is underlain by the Shepparton Formation which consists primarily of silts and clays interbedded with sands and gravels.

During the investigation, two sand/fine gravel layers were identified (upper and lower aquifer) at depths of approximately 10 m and 20 m. Each aquifer ranges in thickness between 2 m and 10 m as shown in Figure 15 and is present over most of the study area.

Groundwater flow in both aquifers is in a general west to north-westerly direction. There is also a general downward flow of groundwater from the upper aquifer to the lower aquifer.

Groundwater quality investigation

Multiple point piezometers were constructed in 150 mm diameter bores at 30 locations across the study area. Bore locations are shown in Figure 14. Groundwater samples were taken from all bores in July 1992 and January 1993 and analysed for the following parameters:

- Nitrate;
- Total Kjeldahl Nitrogen (TKN);
- pH;
- Electrical Conductivity (EC);
- Dissolved Oxygen (DO);
- *E. coli*; and
- Faecal Streptococci

The results of the investigation showed that nitrate was detected in a plume that originated from an area of high septic tank density (>15 septic tanks/km²) centred around bore BH113036 in the Upper Aquifer. The orientation of the plume reflects the general direction of lateral groundwater flow in the Upper Aquifer. A nitrate plume was also found to be present in the Lower Aquifer centred immediately to the west of the plume in the Upper Aquifer. The presence of nitrate in the Lower Aquifer indicates good connection across the clay/silt aquitard most probably through minor sands and gravels. As a result the nitrates present in the Lower Aquifer are most likely a continuation of the plume in the Upper Aquifer. Upper and Lower Aquifer concentrations of nitrate are shown in Figure 15.

Ammonium and TKN were not detected in any monitoring bore at a concentration of more than 1.0 mg/L.

Figure 14
Benalla septic tank study area

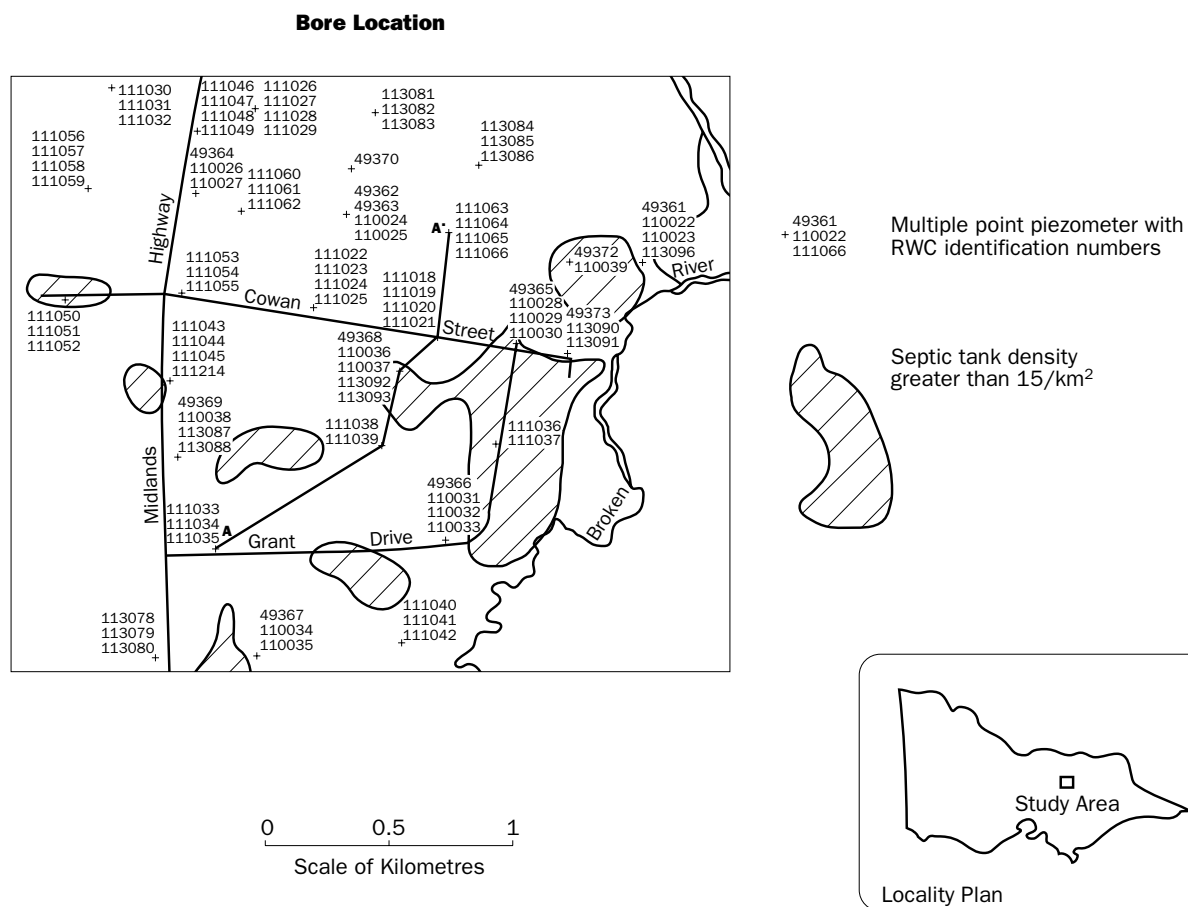
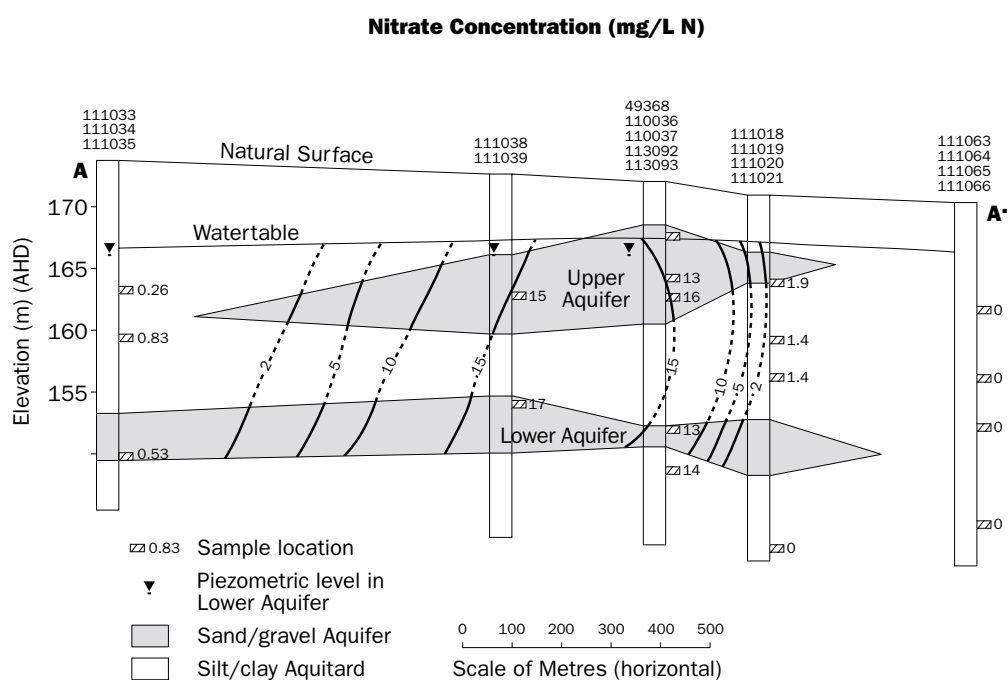


Figure 15
Nitrate concentrations in the upper and lower aquifer



Groundwater contamination processes

The capacity of soil to purify septic tank effluent is dependent primarily on soil composition and hydraulic characteristics. Soil permeability impacts on sorption and chemical reactions by controlling the period of contact between the contaminant and the soil matrix. Soils that are permeable tend to have a lower percentage of clay minerals and reactive species than less permeable soils. This results in a lesser capacity to remove septic tank contaminants in high permeable soils and a higher potential for leaching of nitrate to the watertable.

Due to the high permeability of the soils in this case study, nitrification of ammonium in septic tank effluent will occur relatively quickly. This nitrate is then available to leach to the upper aquifer at a depth of 10 m. The sandy nature of the soils results in low available carbon for denitrification processes to occur, resulting in conservation of nitrate within the aquifer. This conservation of nitrate results in nitrate being available to leach to the lower aquifer.

Conclusions

The Benalla Case Study demonstrates the impact on groundwater from septic tanks in a sandy environment when the septic tank density is above 15/km². Management of the installation of septic tanks should take into account the permeability of the soil and aim to have a septic tank density of less than 15/km².

Nitrate loading from septic tank effluent poses a risk to the beneficial use of groundwater in areas of high septic tank density. In areas where the soil is sandy and there is little available organic carbon, there exists little potential for denitrification to reduce concentrations of nitrate. In this case study, the geology of the aquifer determines the conservation of nitrate and its mobility between upper and lower aquifers.

Natural process nitrate contamination— Yulara, Northern Territory

Introduction

Yulara Case Study is an investigation of high nitrate concentration groundwater from the Northern Territory carried out by Barnes *et al.*, 1992.

The study area at Yulara was initially investigated during the establishment of a tourist resort at the site. High nitrate concentration groundwater was discovered during the initial groundwater investigation for potable water for the development. Thirty groundwater observation bores were installed around the site.

Geological and hydrogeological setting

The study area is underlain by Proterozoic to Cambrian rocks of the Amadeus Basin sequence. Overlying these rocks are Tertiary sediments up to 100 m thick, comprising sand, clay, some calcrete and lignite. Quaternary aeolian deposits form a veneer, and occur either as gently sloping sand plains or as irregular or parallel dunes.

The major aquifers are unconfined or partially confined sand layers in the Tertiary sequence. Fractured bedrock aquifers occur in underlying dolomite, chert and sandstone of the Proterozoic–Cambrian sequence. Groundwater flow is generally to the north-east and discharges to Lake Amadeus, 50 km away. Depth to watertable decreases from 30 m south of the study site to 15 m north of the study site.

Groundwater salinity in the study area ranges from 1,500–5,000 mg/L. Most groundwaters are $\text{Cl} - \text{SO}_4 - \text{HCO}_3$ type with Na the dominant cation with pH ranging from 7.2 to 8.2 and temperature 26°C.

Annual rainfall in the area is approximately 300 mm, with a 30 year record showing the range to be 100–300 mm.

Soil and groundwater investigations

Soil and groundwater samples were recovered from the study site and analysed for nitrate content. Nitrate was found to be present in most bores at a concentration greater than 10 mg/L (as N) with a maximum of 54 mg/L (as N). Highest groundwater concentrations were found in an area where a belt of mulga (*Acacia aneura*) was present.

Soil samples from a number of sources were analysed for nitrate content and the results of the analyses are shown in Table 14.

Nitrate contamination processes

Several potential sources of nitrate to groundwater were identified from soil and groundwater analyses. In soil, significant concentrations of nitrate were identified in:

- young mulga grove;
- spinifex grasses;
- termite mounds; and
- soil impacted by fire.

Table 14
Nitrate concentrations in soil profiles

Site	Range (mg/L)	Median (mg/L)	Maximum (mg/L)
Sand	0.2–11.2	1.3	
Surface Cust	0.5–7.6	1.6	
Bushfire Ash	0.7–6.1	1.8	
Spinifex/grass	1.4–17.2	7.0	
Mulga	1.3–18.2	9.4	
Inactive Termite Mound	5.4–21.8	13.6	
Active Termite Mound	20.2–225	175	
Dune top		1.2	1.0
Mature mulga		15	15
Young mulga		20	7
Spinifex in young mulga		20	18
Bare ground, young mulga		7	22

Low concentrations of nitrate were found in dune tops and on bare ground in mature mulga, and underneath young mulga.

The most significant potential source of nitrate to groundwater was considered to be the termite mounds. Nitrogen fixation occurs in the gut of termites during the digestion of cellulose. It is also hypothesised that bacteria within the mound (free living and gut bacteria) fix nitrogen to ammonia, which is oxidised by other bacteria to nitrate. When rainfall occurs, this nitrate is leached to the ground and infiltrated to the watertable.

The load of nitrate produced by termites in mounds is not able to be quantified due to lack of information on production of nitrate within mounds or mound size, termite numbers and mound distribution.

Natural fire cycles (of approximately 15 years) were also considered to be significant source of groundwater nitrogen. After rainfall, significant concentrations of nitrate were identified in soil close to a fire scar. The fire cycle is proposed to influence recharge and the concentration of nitrate reaching the watertable. Removal of plants by burning causes rainfall that would normally be transpired is available to infiltrate into the sandy soil. Ash from fires is dispersed and the nitrogen is available for leaching.

Denitrifying bacteria are not commonly found in the Australian arid zone due to the lack of soil organic carbon in most soils. Their absence increases the potential for nitrate that is formed to remain conserved in the soil and leach to the groundwater.

Conclusions

Five major conclusions were drawn from this study by Barnes *et al.*, 1992 as follows:

1. Nitrate is fixed in the study area by bacteria. Denitrifying bacteria are scarce and organic carbon in the soil profile is low.
2. Major nitrate contributing areas are termite mounds through the generation of ammonia from nitrogen fixed by bacteria within the mound. This ammonia is oxidised to nitrate by nitrifying bacteria and leached from the mound.

3. Cyanobacteria are prevalent in the dry surface soils and produce nitrate intermittently following rainfall.
4. Recharge pulses following one-in-twenty rainfall events flush nitrate 15–30 m through the soil profile to the watertable.
5. Fire plays an important role in the generation of high-nitrate groundwaters, through the facilitation of recharge and the increased availability of nitrogen at the soil surface.

Reference

Barnes, C.J., Jacobson, G., and Smith G.D. (1992) The origin of high nitrate groundwater in the Australian arid zone. *Journal of Hydrology*. 137:181–197.

Urban land use and mixed agricultural— Perth Metropolitan and Jandakot Mound

CLAUS OTTO, CSIRO LAND AND WATER

PAUL CARLILE, CENTRE FOR GROUNDWATER STUDIES

Definition of urban groundwater

The study of groundwater in an urban environment involves similar concepts to rural applications, however there are certain differences in hydrology and chemistry due to diverse and changing land use. Some of the characteristics which can be seen in urban groundwater, are the effects sewers, stormwater and the sealing of surfaces have on groundwater recharge, with great variability in recharge rates observed over small distances. Tunnels, pilings and basements can result in lowering and rising groundwater.

The urban environment can also restrict groundwater investigations due to the rights of private landholders and intensive land use. Numerous point and multipoint contamination sources such as wastewater systems, parks and gardens, landfills, residential and industrial sources can contaminate urban groundwater. Groundwater management is also complicated by the economic imperatives established in such areas over many years (Lerner, 1996) The evaluation of groundwater contamination by pollutants such as nitrate is dependent on the availability of sufficient high quality data and an understanding of the processes involved.

This paper attempts to identify these processes and investigates the impact of urbanisation on the contamination of groundwater by nitrate by different land uses within Perth metropolitan area, Western Australia. This study is a comprehensive review of available reports and publications and an interpretation of existing groundwater quality data.

Hydrogeology of the Perth metropolitan area

Geology and precipitation in the Perth Basin has resulted in the largest freshwater groundwater resource in Western Australia (Commander, 1990; Davidson, 1995). The regional watertable throughout the basin can be as deep as 100 m on the inland plateau (Darling Scarp) to a few metres or less on the coastal plain. Many groundwater-fed wetlands occur throughout the coastal plain. The superficial aquifer and outcrop areas of the confined aquifer systems form a large unconfined aquifer system on the coastal plain and under Perth, which is at risk of contamination due to the urbanisation and industrial developments. Deeper, confined multi-layered aquifers contain important groundwater resources in the region.

The Swan Coastal Plain is mainly located on Spearwood and Bassendean sands. These sandy soils have poor soil water and nutrient retention capacities, although Spearwood sands are generally accepted to have better retention capacities than Bassendean sands.

Perth's groundwater supply and economic considerations

Groundwater abstraction for the Perth Metropolitan area is primarily from artesian and shallow groundwater from the Gnangara Mound north of Perth and to a lesser degree the Jandakot Mound to the south. Others include Mirrabooka, Gwelup and Pinjar groundwater abstraction sites (Western Australian Legislative Assembly, 1994). The Perth Metropolitan area draws about 40% of its drinking water from groundwater sources located on the Gnangara and Jandakot Mounds. Of this 60% is drawn from production bores in unconfined superficial formations aquifer, which is very susceptible to contamination (Dames & Moore, 1996).

Underground Water Pollution Control Areas (UWPCA; Figure 16), or priority areas, are areas where groundwater is drawn for public water supply. Protection of these areas is the responsibility of The Waters and Rivers Commission. The boundaries of the UWPCA have been defined from “the source areas of groundwater for the production wells within them”, and cadastral boundaries (Dames & Moore, 1996).

The Waters and Rivers Commission has split groundwater areas into three types of priority areas designed to control development above groundwater reserves and protect groundwater quality and quantity. Priority 1 areas are the most important, in that they must be protected if groundwater quality and quantity are to be assured. Priority 1 areas have the highest priority in land-planning management and are generally public owned. Priority 2 areas are designed to ensure there is no increased risk to the groundwater source, while allowing limited development that is managed, in keeping with good groundwater protection practices. The zoning of Priority 2 areas is generally low intensity development (eg. rural), with a maximum septic tank density of one unit per two hectares. Commercial and industrial activities are generally not permitted in Priority 2 protection areas. Priority 3 source protection areas were designed to minimise the risk to the water source. Rather than exclude land uses in Priority 3 areas, groundwater is protected through land use management (Golder Associates, 1996).

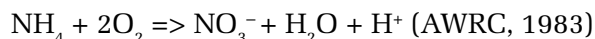
Groundwater abstraction in 1991 within the Perth Area from shallow unconfined aquifers was at a rate of $220 \times 10^6 \text{ m}^3$ per year. This made up 75% of the total groundwater abstraction within the Perth Basin. The rapid population growth taking place in Perth requires that further urban residential development takes place. Much of this development is taking place north of Perth and is resulting in land-use changes. These changes often result in encroachment on natural bushland. As such, developments move further north and south they also leave behind progressively older urban environments. These older urban environments have been shown to have greater nitrate concentrations than newer urban areas, or native bushland (Appleyard, 1995).

Proposed water source development in Perth to the year 2010 includes 85% from groundwater. It is estimated groundwater will contribute 50% of all water supplied to the Perth region by this time (Stokes *et al.*, 1996). At present, groundwater from both superficial and confined aquifers on the coastal plain contribute 100 GL/yr or 35% of the total water supply to Perth and the Goldfields. Stokes *et al.* (1996) points out that if all local groundwater reserves became contaminated additional treatment costs will range from “20 cents/kilolitre for ion exchange processes to remove nitrate, to 60 cents/kilolitre for an activated carbon process to remove multiple contaminants” the future cost to remedy the problem could be between \$600 and \$1,800 per household. Contamination would also effect private bores which presently supply 50% of Perth’s total water needs (Stokes *et al.*, 1996). Slow movement of groundwater within many Perth aquifer systems also means that some aquifers may remain contaminated for decades, before emerging in water supplies, thought to be suitable for public supply.

Nitrate source processes and land use

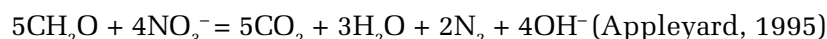
A review of the literature available on nitrate levels in groundwater aquifers within the Perth urban region shows that **groundwater nitrate levels are increasing**. Many reports present conflicting groundwater nitrate concentrations for the same area, however one clear pattern does emerges. Leachate nitrate concentrations are generally higher from sites on Bassendean sands, while groundwater nitrate concentrations below Bassendean sands are generally lower. Spearwood sands generally have low leachate concentrations and high groundwater nitrate concentrations (Sharma *et al.*, 1993). This pattern is not always followed and high groundwater nitrate concentrations have been found under both sand types. Presently Australian drinking water guidelines listed in the NH&MRC (1996) are less than 0.5 mg/L of ammonia as ammonium or 0.4 mg/L as N, nitrate of 50 mg/L as nitrate or 11.3 mg/L as N and nitrite of 3 mg/L as nitrite and 0.9 mg/L as N. Nitrate levels exceeding 11.3 mg/L as N have been shown to produce adverse health effects in people, especially children. Therefore any groundwater with nitrate levels higher than this, is cause for concern.

Nitrate content in groundwater is related to both the source of nitrate and the nitrogen cycle. The Australian Water Resources Council (AWRC, 1983) has described nitrate contamination in Australia and the various sources of nitrate contamination. Groundwater nitrate contamination not only includes man-made sources, but also natural sources. Organic matter is a source of nitrate in which decomposition of plant, animal and microbial decomposition adds to soil nitrogen. Nitrification of ammonium (from a variety of sources) to nitrate, resulting in nitrate leaching to groundwater. Nitrification is an aerobic reaction that takes place in moist conditions and can involve various stages. For example, one species of bacteria may oxidise ammonia to nitrite, and another oxidises nitrite to nitrate, with the overall reaction being similar to :



Ammonia can also be assimilated through the process of nitrogen fixation, whereby atmospheric nitrogen is fixed, most commonly by legume plants that have symbiotic relationships with nitrogen fixing bacteria. The fixed nitrogen in the form of ammonia is then added to the organic matter in the soil before possibly being involved in nitrification, and leached to the groundwater as nitrate. Other sources of nitrate contamination of groundwater identified in the AWRC report include commercial fertilisers, animal wastes, sewage, municipal and industrial wastes, precipitation, and geologic sources. The Perth Basin has been reported to have nitrate rich groundwaters, and there are various studies which would agree with this prognosis.

The increase in urbanisation within the Perth metropolitan area has resulted in increased nitrate concentrations within the surficial aquifer. The increases are however, often lower than expected. This can be explained by the process of denitrification in which “heterotrophic bacteria in the presence of organic carbon, reduce nitrate to nitrogen gas, through reactions such as:”

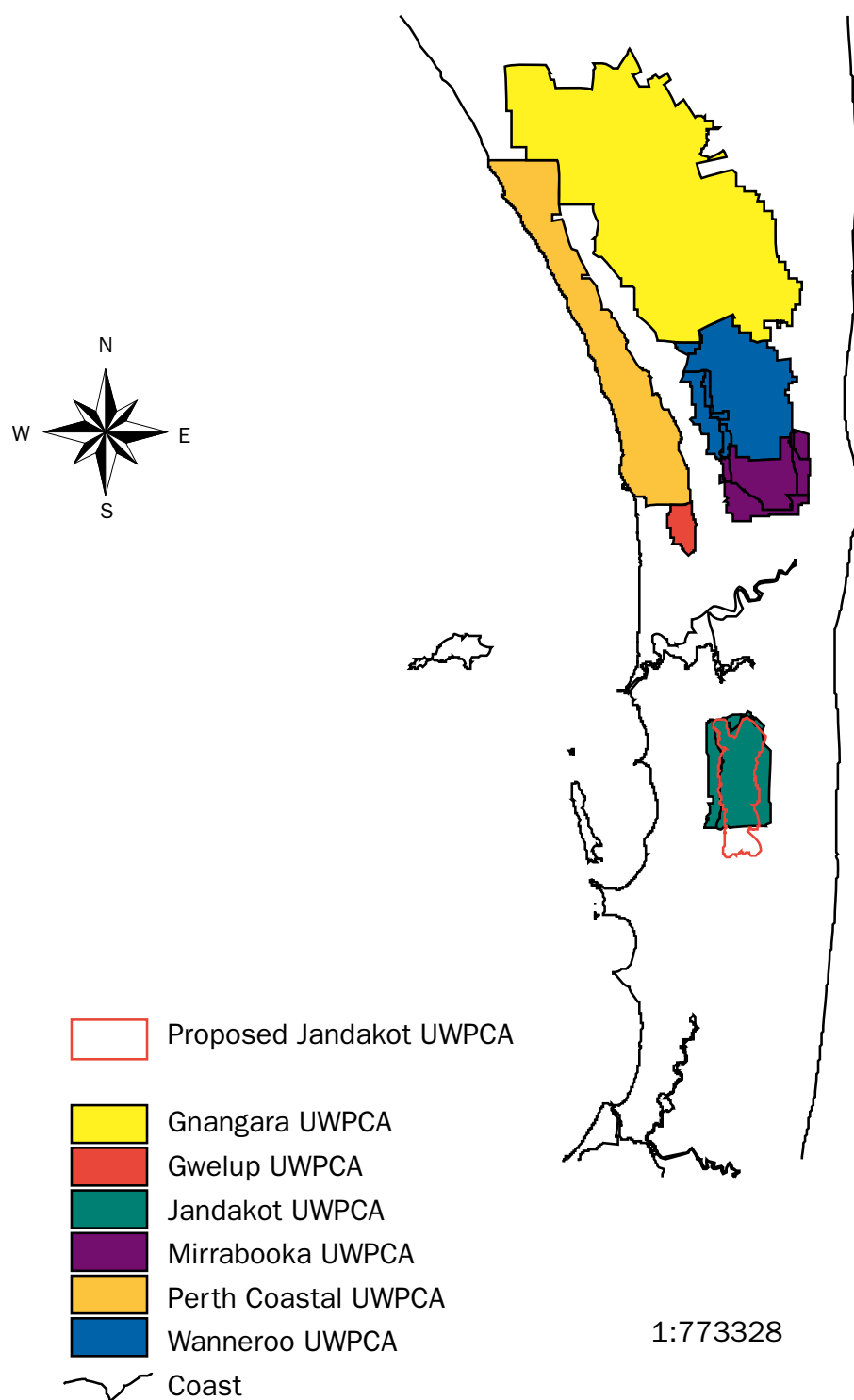


When organic carbon is absent however, chemicals such as sulphides and Fe(II) minerals can reduce nitrates to nitrogen. The unconfined Superficial Tamala Limestone formation of north coastal regions of Perth is generally low in all such reducing agents, and the limited supply of such agents may result in future increases in groundwater nitrate concentrations, however further research is recommended.

The following is a summary of the work done within the Perth Metropolitan Area relevant to nitrate contamination processes. The studies are presented in chronological order in the hope of identifying trends.

Whelan and Parker (1981) examined groundwater pollution from a septic tank in a Bassendean Sand at a site in a north eastern suburb of Perth. A large number of septic tank systems close to open drainage channels are found in high watertable, Bassendean Sands. Bassendean Sands generally have a low retention capacity for nitrogen and phosphorus within the profile. Whelan and Parker (1981) found that these chemicals enter the groundwater rapidly and at the same concentration as in the effluent. It was recommended that priority be given to sewerage those urban areas with Bassendean Sands and high watertables. It was also found that at one end of the leach drain conditions were saturated and anaerobic, thereby preventing the oxidation of ammonia. This meant that at one end of the leach drain, nitrogen was present as ammonia, and at the other end as nitrate. Groundwater nitrate concentrations were found to be as high as 70 mg/L in some places.

Figure 16
Location of the Perth groundwater priority protection areas



In 1984 Whelan and Barrow studied the transformation of nitrogen under septic tanks in sandy soils (Spearwood and Bassendean) on the Swan Coastal Plain. Household toilets were found to be the main contributors of nitrogen in septic effluent, mainly in the form of ammonium. Once the effluent passed the slime layer of the absorption systems into the aerobic soil beneath, it was oxidised to nitrate within 0.5 m. Occasionally this zone between the slime layer and the watertable was saturated and anaerobic, resulting in oxidation of ammonium to nitrate not taking place. Generally though the aerobic conditions and low cation exchange capacity of the soil resulted in nearly all oxidised ammonium (ie. nitrate) entering the groundwater or being taken up by plants.

In 1984 almost half of the households in Perth used septic systems for effluent disposal. These septic systems comprised a concrete tank “which act as anaerobic digestion vessels”, and then release the treated effluent into soak wells or leach drains. Whelan and Barrow (1984) reported that water flow from these systems and generally throughout most of the Swan Coastal Plain is vertical, with most of the effluent waste reaching the groundwater. It was pointed out, however, that groundwater used as drinking or public supply was unlikely to be effected by septic tank effluent.

Gerritse *et al.* (1988) investigated the “*effect of urbanisation on the quality of groundwater in Bassendean sands*,” along a north-south transect in Perth, Western Australia. In general Bassendean sands were found to remove nitrates effectively from the superficial aquifer, due to the favourable conditions for denitrification. This was found not to be the case for Spearwood sands where groundwater nitrate levels were typically higher and conditions were not favourable to microbial denitrification (Gerritse *et al.*, 1988). Each Perth household consumes on average 18 kg N/year, most of which ends up in sewage or septic effluent. Council parks received approximately 150 to 200 kg N/ha, while broad acre grasslands received 30 to 40 kg N/ha. Gerritse *et al.* (1988) reported that nitrogen application in sewered urban areas was about 80 kg N/ha compared to 260 kg N/ha in unsewered areas. Nearly 50% of nitrogen from fertilisers in urban areas leaches to the groundwater. This would amount to groundwater nitrate concentrations of 40 mg/L in sewered areas, as compared to 70 mg/L in unsewered areas. This was found not to be the case under Bassendean sands, however, most probably due to intensive microbial denitrification. This hypothesis was substantiated by high organic matter concentrations in the groundwater and a pH range of 5 to 7, which is ideal for microbial growth. Groundwater from below Bassendean sands were also found to have redox potentials well under 300mV making them ideal conditions for denitrification (Gerritse *et al.*, 1988).

Concentrations of all major ions in the superficial aquifer were found to increase with the level of urban development. Gerritse *et al.* (1988) expressed concern that without accurate data on rates of nutrient input, calculating the potential ‘breakthrough’ of these nutrients into the groundwater is made impossible.

A Western Australian Department of Agriculture survey of groundwater nitrate concentrations from 40 bores in market gardens found that over half the bores had groundwater nitrate concentrations which exceeded the World Health Organisation Guideline of 10 mg/L $\text{NO}_3\text{--N}$. It was also found that nitrogen rich groundwater used for irrigation adds to crop nitrogen supply and this requires that growers estimate irrigation water nitrate concentrations, in order to use fertiliser applications more sparingly (Lantzke, 1995).

Contaminated run-off could also be a source of nitrate contamination of groundwater. Run-off can accumulate in low-lying areas and infiltrate through sediments to groundwater. Tan (1991) presented results for nitrate, nitrite, total nitrogen and ammonia concentrations in run-off from suburbs in Perth including areas on Bassendean and Spearwood sands, with both sewered and septic systems. All run-off concentrations were below 3 mg/L and appeared not to play a major role in contamination of groundwater.

Sharma *et al.* (1992) found that leachate beneath the root zone of horticultural activities are comparable to that found in the leachates of septic tanks and sewerage effluent. Fertiliser application within the Perth area have been estimated at around 5,000 tonnes/yr, which provides approximately 400 tonnes of nitrogen that can be leached to the groundwater and finally end up in the Swan–Canning and lake systems (Sharma *et al.*, 1992).

Davidson (1995) found that intensive horticultural and urbanised areas of the Perth Region have resulted in high nitrate levels in the superficial aquifers. This is generally the result of leaching of nitrate from fertilisers, although reduced nitrogen compounds such as ammonia, may undergo nitrification by bacteria to form nitrite and nitrate. Forested, rural and native bushland areas of the Perth Region generally have groundwater nitrate levels less than 1 mg/L. These areas reflect the low nitrogen input by the land-use. Legumes such as *Acacia* along the coastal limestone belt of Perth fix nitrogen and the higher soil nitrate concentrations are reflected in the groundwaters. Intensive poultry, pig farming and the disposal of industrial wastes have resulted in high nitrogen concentrations locally. Market gardening within the urban area has resulted in groundwater nitrate concentrations exceeding 20 mg/L. Groundwater beneath fertilised gardens can have nitrate concentrations up to 20 mg/L. Areas including horticulture, recreational grounds and septic sewage had groundwater nitrate concentrations between 20 and 60 mg/L, while nitrate concentrations greater than 60 mg/L were attributed to industrial wastes.

Davidson (1995) regarded the Pinjar, Wanneroo, Mirrabooka and Jandakot water schemes to be “unaffected by urban nitrate sources.”

Using suction driven lysimeters Sharma *et al.* (1996) measured water and nutrient fluxes beneath the root zone of public and private lawns in urban areas of Perth, Western Australia. The sites received similar fertiliser applications, but irrigation was different from site to site. It was reported that nitrate annual flow weighted concentration in the leachate was between 0.8–5.4 mg/L and often exceeded 10 mg/L. Sharma *et al.* (1996) expected groundwater nitrate concentrations would be below the World Health Organisation drinking water limit of 10 mg/L, except for short periods of time and that there was a greater threat to wetlands into which nutrient rich groundwater discharges. As more areas become sewerred, fertilised gardens and lawns will then become the major sources of nitrate to groundwater, due to an ever increasing population.

Lantzke (1997), examined nutrient loss from nine horticultural sites on sandy soils of the Swan Coastal Plain. Six of these sites had monitoring bores installed and monitored for phosphorus (P) and nitrate ($\text{NO}_3\text{--N}$) levels. Most of the properties were located on Bassendean sands. Beneath production areas of all properties shallow groundwater $\text{NO}_3\text{--N}$ concentrations were high (> 10 mg/L) to very high (> 50 mg/L), however these elevated $\text{NO}_3\text{--N}$ concentrations were generally limited to the upper part of the superficial aquifer. High nitrate concentrations were also found up to 100 m down gradient of four of the six horticultural production areas (Bassendean sands). The other two horticultural properties (Joel sands) showed decreases in nitrate concentrations in shallow groundwater; with increasing distance from both properties. These two lower results were explained by the denitrification process. The Lantzke (1997) report includes nitrogen and phosphorus application rates among the nine horticultural properties, time line series nitrate data (1994–95 and 1995–96) and maximum groundwater nitrate concentrations beneath various horticultural activities. These groundwater nitrate concentrations ranged from 7 to 110 mg/L nitrate as N ($\text{NO}_3\text{--N}$).

The Waters and Rivers Commission report into the Canning Vale and Banjup Kennel zones, considered the burial of faeces inappropriate, because of the resultant nitrogen ‘hot spots’. These are prone to leaching and could be detrimental to public health (Waters and Rivers Commission, 1997). Here denitrification was estimated by to be at a rate of 10% however from a typical 2 ha special rural lot in priority 2 areas. Measured results, however, were typically lower (although still elevated) than the predicted result and it was assumed that the rate of denitrification within the two kennel zones must have been underestimated (Waters and Rivers Commission, 1997).

Groundwater contamination by nitrate in Perth's urban areas

Various studies in the Perth urban environment have been carried out at specific sites (including Gnangara, Jandakot and Gwelup), to investigate groundwater nitrate levels and contamination processes below a range of land uses. Bestow (1981) presented data for groundwater quality beneath a land-fill site within the Gwelup water supply scheme. Nitrate levels ranged from 0–1 mg/L, while ammonia levels ranged from 0–190 mg/L. Bestow explained that the rising ammonia levels could be due to “leakage of ammonia through the peat and clay,” due to reduction in adsorption capacity, or because there was insufficient oxygen for nitrification (conversion of ammonia to nitrate).

The high solubility of nitrate and its relative inability to be held by soils allows it to be readily leached to groundwaters. Maring and Harris (1982) examined groundwater physical and chemical properties of the Gwelup Wellfield (Spearwood sands) and the Mirrabooka Wellfield (Bassendean sands). It was suggested the higher nitrate values found at the Gwelup (partly unsewered) was linked to the high ferrous iron content which interacts with the redox potential through the “ferrous-ferric couple rather than the nitrogen-nitrate couple.”

There are however conditions at the Mirrabooka site which make it more suitable for denitrification of NO_3^- to N_2 . These conditions are low redox potential, high organic carbon, suitable pH (5.0 and 7.0) and anaerobic conditions. This study shows the differences between a closed system at Mirrabooka where oxidizing species (O_2 , NO_3^- , SO_4^{2-} and CO_2) and excess organic carbon enter the system at groundwater recharge points before the groundwater system becomes closed and therefore anaerobic (conditions which are favourable to denitrification). The Gwelup groundwaters exhibit an open system where the increased recharge area has resulted in excess dissolved oxygen and suitable conditions for nitrification rather than denitrification (Maring and Harris, 1982).

Both the Applecross and Gwelup flow lines were examined vertically for nitrate concentrations. At the Gwelup flow line (northern Perth Area) groundwater nitrate concentrations were found to be between 1 mg/L and 29 mg/L. At the Applecross flow line (southern Perth Area) groundwater nitrate concentrations were low, beneath native bush and new urban area. However older urban areas all had elevated nitrate concentrations, probably due to high septic tank density, restricted throughflow due to location within the Applecross Peninsula and because of blending of groundwater within the aquifer caused by ‘heavy private pumping’ (Water Authority of Western Australia, 1987). Included in the Water Authority of Western Australia (1987) report are land-use maps, and maps showing septic tank density and horticultural enterprises. Bore sampling locations within the Gwelup and Applecross flow lines are shown including vertical nitrate concentrations.

Pionke *et al.* (1990) investigated groundwater nutrient concentrations and root zone drainage at Coogee and Gnangara, on Spearwood sands within the Swan Coastal Plain, Western Australia. Measurements were taken from lysimeter equipped sites and shallow and deeper irrigation wells. The average nitrate concentration at the Coogee site from irrigated wells was 31.2 mg/L NO_3^- -N. The Gnangara (deep groundwater) irrigation wells were found to have an average nitrate concentration of 10.2 mg/L NO_3^- -N. Nitrate concentrations in root zone drainage beneath market gardens ranged between 71 and 209 mg/L. It was found that poultry manure applications exceeded crop uptake by five times for nitrogen at many sites, but denitrification was taking place at the Gnangara site. Here the denitrification process simply relied on the depth of sand above the groundwater to supply anaerobic conditions.

Barber *et al.* (1991) investigated leaching of nutrients, such as nitrogen and phosphorus, in urban Shire Council parklands environments, with known rates of fertiliser application. It was found that nitrate moved easily through the unsaturated zone of both the Bassendean and Spearwood Sands. Extensive denitrification occurred in groundwater in shallow anoxic Bassendean Sands and in Spearwood Sands where the groundwater was within 3–4 m of the surface.

Natural clean-up of groundwater by denitrification was found to be clearly related to the redox status of the groundwater. They recommended that more information be gathered on redox potentials of the Swan Coastal Plain so that “regional assessment of the relative vulnerability to pollution by organic compounds and nitrate”, could be assessed.

Sharma *et al.* (1991) reported on the “impact of horticulture on water and nutrient fluxes to a sandy aquifer,” within the Gnangara Mound. This included measurement of nutrient and water balances at two market gardens, and groundwater nutrient concentrations on site and downstream from these properties. At farm A groundwater nitrate concentrations ranged from 80 mg/L beneath the farm to 0.1 at downstream wells and indicated that little nutrient was being transported away from the farm in groundwater. At Farm B groundwater nitrate concentrations ranged from 75–20 mg/L.

Sharma *et al.* (1992) investigated nutrient fluxes beyond the root zone from the lawns of two urban areas; Karrinyup (Spearwood sand) and Mount Lawley (Bassendean sand) and two parkland sites; Noranda (Bassendean sands) and Tuart Hill (Spearwood sands), Perth, Western Australia. Fertiliser application rates were determined for all sites as well as the nutrient composition of irrigation water, precipitation and public water supply. The Tuart Hill site had no fertiliser applied, although about 30 kg/ha of nitrogen was applied through irrigation water pumped from groundwater. At the other three sites fertiliser application ranged from 80 to 235 kg/ha during the investigation.

The two Bassendean sand sites showed higher nitrate leachate concentrations than the two Spearwood sites. Spearwood nitrate leachate concentrations ranged from 1 mg/L to 10 mg/L. While Bassendean sites ranged from 1mg/L to 45 mg/L $\text{NO}_3\text{-N}$. Leachate flow-weighted $\text{NO}_3\text{-N}$ concentrations for the four sites were ranked from lowest to highest, such that Karrinyup < Tuart Hill < Mt Lawley < Noranda, with nitrate concentrations of 1.46, 4.20, 10.04, and 10.28 mg/L respectively (Sharma *et al.*, 1992). Time line data is presented from 1991–92 as well as graphs of expected Vs observed nitrate leachate concentrations. From this data it has been estimated that 50% of groundwater nitrate concentrations beneath Perth lawns exceed the World Health Organisation (WHO) safe drinking limit of 10 mg/L (Sharma *et al.*, 1992).

Sharma *et al.* (1993) followed up the 1992 study by including four additional sites. Nitrate leachate concentrations from these areas were as follows. Ballajura (private lawn, Bassendean sand) 5.37 mg/L, Balcatta (public park, Spearwood sand) 4.04 mg/L, Corderoy Reserve (public park, Bassendean sand) 5.33 mg/L and Watermans (private lawn, Spearwood sands) 0.83 mg/L. Of the original sites included in the 1992 study, Karrinyup was the only site to show increases in nitrate leachate concentrations while the other sites (Noranda, Tuart Hill, and Mt Lawley) showed decreases. It is clearly seen that leachate nitrate concentrations are generally higher from sites on Bassendean sands, while groundwater nitrate concentrations are generally lower when compared to Spearwood sands (Sharma *et al.*, 1993).

A study done by Barber *et al.* (1993) found that urbanisation of the “Gwelup Underground Water Pollution Control Area (UWPCA)” showed elevated levels of nitrate in groundwater approaching the current drinking water standard. It was also expressed that it could take some 30 years after the start of urban development for the full impact of unsewered areas to take effect.

Appleyard (1995) found that nitrate concentrations in groundwater under uncleared native vegetation (Barragoon) was half the concentration under two other urban areas (Whitfords and Nedlands). Gerritse *et al.* (1990) evaluated nitrogen application in sewerred urban areas in Perth to be up to 80 kg/ha. Assuming this amount reaches the watertable, mean groundwater nitrate concentrations should be approximately 40 mg/L as N. Appleyard (1995) however reported nitrate concentrations in two urban areas of Perth to be in the range of 0.1–5.6 mg/L, and suggested denitrification accounted for the lower than expected concentrations within the aquifer. Appleyard (1995) also found that urban areas had higher groundwater redox potentials than non-urban areas and pointed out that this may reduce denitrification, which only occurs at redox potential's less than about 300 mV.

The investigation into leaching of nutrients beneath urban lawns within the unconfined sandy aquifer occupying much of the Swan coastal plain was continued and presented in the progress report by Sharma *et al.* (1995). This ongoing study of four sites (Tuart Hill, Karrinyup, Balcatta, and Waterman) situated on Spearwood sands and four sites (Corderoy, Mt Lawley, Ballajura and Noranda) located on Bassendean sands examines the nutrient concentrations within the leachate beneath the root zone, and nitrate input concentrations including irrigation water. Nitrate applications in the form of fertiliser and irrigation take place at all sites predominantly in summer. It was found that high irrigation rates resulted in greater leaching and that reductions in nitrogen leaching could be achieved not only by reductions in fertiliser, but also by reductions in irrigation. Again most sites had $\text{NO}_3\text{-N}$ concentrations approaching the safe drinking water limit. The leachate for Bassendean sands was again found to have higher nitrate concentrations than Spearwood sands, although it was pointed out that soil type might not be the only contributing factor, and that the higher recharge rates of Bassendean sands may also be responsible. Flow weighted mean concentrations of leachate over the period of a year were less than 5 mg/L at all sites. Public sites had much lower nitrogen input than private sites (130 kg/ha compared to 350 kg/ha) (Sharma *et al.*, 1995).

Land-use changes in the Gwelup study area have seen a steady decrease in natural bushland, horticulture and septic tanks, while parklands, and sewered areas have increased. The highest concentrations of nitrate were found in the southeastern area where unsewered areas were developed in upgradient areas (post-1960 before houses were sewered in the mid-1970s). Surprisingly it was found that nitrate was largely derived from unsewered areas rather than horticulture in the area. Again denitrification in peaty soils, was seen as the reason for reduced nitrate impacts from horticultural areas. Overall, Barber *et al.* (1996) concluded that “local flow regimes within the unconfined aquifer delineate oxidising (nitrate rich) areas from more reducing (denitrifying areas)”, and this has reduced the impact of areas previously used for horticulture.

Managing nitrogen for groundwater quality on the Jandakot Mound, Perth

The Jandakot mound 30 km south of the City of Perth and the Swan River is an important groundwater abstraction and recharge area. The crest of the mound is protected by a UWPCA and declared a Priority 2 and 3 area. The UWPCA is mainly underlain by Bassendean Sands and the Superficial aquifer. Land uses on the Jandakot mound are residential, rural smallholdings (eg. kennel zones), horticulture, floriculture, intensive animal industries, industrial and commercial activities, parks and ovals, and remnant bushland.

The Jandakot mound is located within the Peel–Harvey coastal catchment and therefore land uses objectives here impact not only on groundwater quality, but also on eutrophication of the Peel–Harvey Estuary. Various planning considerations have been made to protect this valuable resource from degradation in quality and quantity. The EPA still holds the view that “urban development on land above the Jandakot groundwater mound between the two lines of public water supply bores” is inappropriate, and the area should be included in the rural landscape and conservation area. The area between the bore lines should remain exempt from any urban development and zoning could be changed from rural to Special Rural to give local authorities greater controls on land use (EPA WA, 1993). The Environmental Protection Authority Bulletin No. 587 (EPA WA, 1991) describes the Jandakot groundwater scheme Stage 2. It discusses several land use proposals for the Jandakot mound, but there is little or no discussion on groundwater quality, including nitrate levels.

The EPA (EPA WA, 1993), considered the Jandakot Land-Use and Water Management Strategy and recommended that larger rural block sizes on the Jandakot Mound to reduce the impact of population density and on-site effluent disposal and allow better control of clearing. The EPA considers that for effluent disposal systems to work properly; the bottom of the leach drain must be “a minimum of 2 metres above the highest watertable and 100 metres from the nearest water body or drain.” Therefore on site effluent disposal systems often require a height of fill to be used so the depth to the watertable is increased and the risk of groundwater contamination is decreased. The strategy also pointed to the impact of grazing animals and recommended restrictions on the number of animals per property, such as one horse (or stock equivalent) per 2 hectare lot.

Overall rural land uses should be restricted to “broad acre, dry land grazing” to reduce the impacts to both the Jandakot Mound groundwater quality and nutrient input to the Peel–Harvey estuary (EPA WA, 1993). Market gardens and horticulture were also seen as environmentally unacceptable due to the irrigation requirements and high nutrient inputs that would take place on sandy soils of the Peel–Harvey coastal plain catchment. Likewise intense animal production, dairy, aquaculture and irrigated fodder production were seen as unacceptable land uses on the Jandakot Mound, particularly the Priority 2 source area. The EPA also recommended that commercial and industrial land uses that could pollute the groundwater, be restricted (EPA WA, 1993).

Groundwater nitrate concentrations within the Jandakot Mound and surrounding areas have generally been found to be low. In 1991 the Water Authority of Western Australia found groundwater nitrate concentrations within the Jandakot Mound to be below 0.1 mg/L. Davidson (1995) reported groundwater nitrate concentrations less than 1 mg/L within Jandakot groundwater resources. Golder Associates (1996) investigated water quality from two kennel subdivisions, in Jandakot and found groundwater nitrate levels as high as 9.7 mg/L, however most monitoring showed lower levels of nitrate in groundwater. We investigated 204 bores in the Jandakot area and found that the nitrate levels in groundwater can range from a low a 0.1 mg/L to as high as 100 mg/L in certain areas (see next chapter on nitrate trends).

The Water Authority of Western Australia (1991) reported that in the Jandakot PWSA there are 22 sources of animal based waste, two abandoned waste land-fill sites and one source of industrial waste.” Of these it was said that the industrial site was the only source of groundwater contamination and this is located downstream from all wells of the PWSA, so was considered not to effect groundwater quality. Groundwater nitrate concentrations within the Jandakot mound were found to be generally below 0.1 mg/L.

Golder Associates undertook groundwater quality investigations at Kennel subdivisions Jandakot, Western Australia. Septic-tank densities in the Canning Vale kennel subdivision were found to exceed one unit per two hectares and commercial activities such as soil/manure blending practices were both found to be inconsistent with Priority 2 source protection area guidelines. Water was sampled from various bores (public and private) and tested for nutrient content, in order to evaluate the level of groundwater contamination within the subdivision.

The Kennel subdivisions of the Jandakot area included the Banjup Kennel Subdivision (ten licensed kennels), and Canning Vale Kennel subdivision (75 licensed kennels). Both areas have been 75% cleared and waste disposal is usually ground burial, disposal in domestic septic tanks and off-site disposal. The numbers of dogs range from 7 to 50 per hectare. All properties within both subdivisions are “connected to individual septic tank and leach drain systems.” (Golder Associates, 1996).

The public monitoring bores showed low levels of “ammonium (≤ 0.55 mg/L as N), nitrate (≤ 4.6 mg/L as N), nitrite (≤ 0.02 mg/L as N), and total nitrogen (≤ 4.9 mg/L as N).” It was reported that elevated nitrate levels were recorded in bore number J140 in April 1979 and J141 in 1996. The private bores monitored showed low nitrate levels ranging from 0.50 to ≤ 0.02 mg/L, and nitrogen as N levels between 0.14 to 2.1 mg/L. Higher ammonia levels, however were found in five private bores within the Canning Vale subdivision. The drilled monitoring bores at the Canning Vale subdivision showed low levels of ammonia, nitrate, and nitrite, while those at the Banjup Kennel subdivision showed higher levels of nitrate contamination (≤ 9.7 mg/L as N) at 2.9 to 6.0 m below the watertable, however most monitoring bores showed lower levels. Any higher levels of nitrate recorded at both sites were determined to be of local origin.

Onsite burial of animal faeces disposal of septic wastes were or are contributing to increased nutrient load from each property and there was found to be a “potential for migration of these contaminants to nearby Water Corporation production bores” (Golder Associates, 1996). Concern was also raised that the on-site septic waste disposal practices of the kennel subdivisions and the activities of nearby soil-blending facilities within the Jandakot Priority 2 Source Protection area were not consistent with the Priority 2 Source Protection area guidelines.

The Centre for Groundwater Studies in Perth (pers. com. Claus Otto) is currently investigating the vulnerability of groundwater contamination by nitrate from different land uses on the Jandakot Mound. A honors thesis by Narah Stuart (1997) has recently been completed which applies a mass balance equation coupled with a groundwater flow model to determine groundwater contamination by nitrate from different land use on the Jandakot Mound. This thesis has also identified poor understanding of denitrification processes on the Swan Coastal Plain and lack of loading rates from different land uses, especically intensive animal industry.

Trends in nitrate contamination and groundwater monitoring

To identify trends in nitrate contamination of groundwater requires consistent quality data, at a regular sampling interval, at the same locations. Unfortunately most of the studies examined here have obtained their results from different locations or bores, over relatively short periods of time. Rarely have any of the studies monitored from the same bore locations; so direct trends in aquifer nitrate concentrations are difficult to identify. In some cases groundwater nitrate concentrations have not been measured at all. Instead the concentration of the leachate is given before it undergoes any potential transformation within the aquifer. These leachate measurements do however give an insight into the concentration of effluent after it passes through the root zone of various media, and is therefore an indication of the nitrate input to Perth groundwaters. There have also been trends or patterns identified over the years that correlate soil type and depth to groundwater to nitrate concentrations and the nitrogen cycle. Bassendean sands for example have been commonly associated with denitrification of nitrate within the aquifer.

The Waters and Rivers Commission is responsible for monitoring groundwater quality. Unfortunately monitoring of groundwater nitrate levels within water protection areas, is often too infrequent to supply convincing trends in nitrate contamination. The reason for this could be the cost of monitoring and the assumption that nitrate groundwater levels change slowly with time. Various authors have found problems gathering water quality data from government departments. Dames & Moore (1986) reported water quality monitoring to be infrequent, due to the assumption that groundwater water quality changes less rapidly than groundwater water levels. They therefore proposed that the existing water quality monitoring program (carried out by the then Water Authority of Western Australia) continue without any apparent changes. The EPA (EPA WA, 1987) includes a recommendation that the “Water Authority continue to review and develop methods to improve monitoring and control of all public and private bores, for the purpose of managing the water resource” of the Gnangara Mound. Also that the Water Authority should submit “brief annual and more detailed triennial reports on environmental monitoring and management of the Gnangara Mound to the EPA.”

The Perth Urban Water Balance Study (Water Authority of Western Australia, 1987) found previous monitoring of groundwater quality by the Water Authority of Western Australia, was limited to a few selected bores that made regional assessment of groundwater quality difficult. The report recommended that for management of the urban water balance and/or quality to be effective, the Water Authority must monitor groundwater quality and quantity on a regular basis, so that future predictions are accurate, and so groundwater of the Perth region is managed effectively. Hirschberg (1991) also reported problems gathering data, as the data was spread over various government departments and was in various formats, making comparisons very difficult.

With these problems in mind, this section attempts to identify trends by putting the investigations of groundwater nitrate concentrations reviewed here, into common suburbs or zones, that may or may not be part of groundwater protection areas.

Various studies of groundwater nitrate concentrations over the years have identified specific soils within urban areas of Perth that exhibit various stages of the nitrogen cycle, and allow passage of contaminants to the groundwater to varying degrees. La Brooy (1981) considered Bassendean Sands as one of the worst soils for protecting groundwater from pollutants, because of their lack of organic matter, silica nature and the close proximity of the watertable. La Brooy regarded Spearwood Sands as having a high ability to buffer groundwater from contamination, due to their slight clay content, and increased depth to groundwater. La Brooy regarded Perth soils in general as poor at retaining nitrate with a genuine risk of groundwater contamination in most areas of the Swan Coastal Plain. Wells *et al.* (1986) supported the view that soils of the Spearwood Dune system are suitable for supporting most urban and rural-residential development. The deep, permeable nature of the soils makes them suitable for on-site effluent disposal, whereas most of the Bassendean Dune system is not suitable for on-site effluent disposal (Wells *et al.*, 1986).

In comparison, Whelan and Barrow (1984) found little difference between nitrate leachate concentrations beneath Spearwood and Bassendean sands when studying the transformation of nitrogen under septic tank installations in sandy of sites on the Swan Coastal Plain. Generally the aerobic conditions and low cation exchange capacity of all soils resulted in most oxidised ammonium (ie. nitrate) entering the groundwater or being taken up by plants. Many other authors however have pointed to lower groundwater nitrate concentrations under Bassendean sands when compared to Spearwood sands due to the process of denitrification.

Mulvey *et al.* (1986) found groundwater nitrate concentrations beneath Burswood Island Resort Development (Bassendean sands) to be generally low and considered construction here not to have a detrimental effect on groundwater quality. Gerritse *et al.* (1988) found Bassendean sands along a north-south transect in Perth, Western Australia, to remove nitrates effectively from the superficial aquifer, due to the favourable conditions for denitrification. This was not the case for Spearwood sands, in which groundwater nitrate levels were higher. Gerritse *et al.* (1988) also found Bassendean sands had relatively low groundwater nitrate concentrations ranging from 0 to 0.03 mg/L. In 1990 Gerritse *et al.*, again examined the “impact of residential urban areas on groundwater quality” and found Bassendean sands to have much lower groundwater nitrate concentrations than expected (based on nitrate inputs), markedly lower than results from groundwater beneath Spearwood sands. In 1990, Pionke *et al.*, found that denitrification was taking place in some of the deeper wells at the Gnangara site.

Barber *et al.* (1991) investigation into the leaching of nutrients, such as nitrogen and phosphorus in urban parkland environments, found that nitrate moved easily through the unsaturated zone of both the Bassendean and Spearwood Sands. Although extensive denitrification was found to occur in groundwater under Bassendean Sands and in Spearwood Sands where the groundwater was within 3–4 m of the surface. Nitrate values in the groundwater beneath parklands ranged from 0–8 mg/L in Bassendean Sands and 0–50 mg/L in Spearwood Sands. Natural clean-up of groundwater by denitrification was found to be clearly related to the redox status of the groundwater.

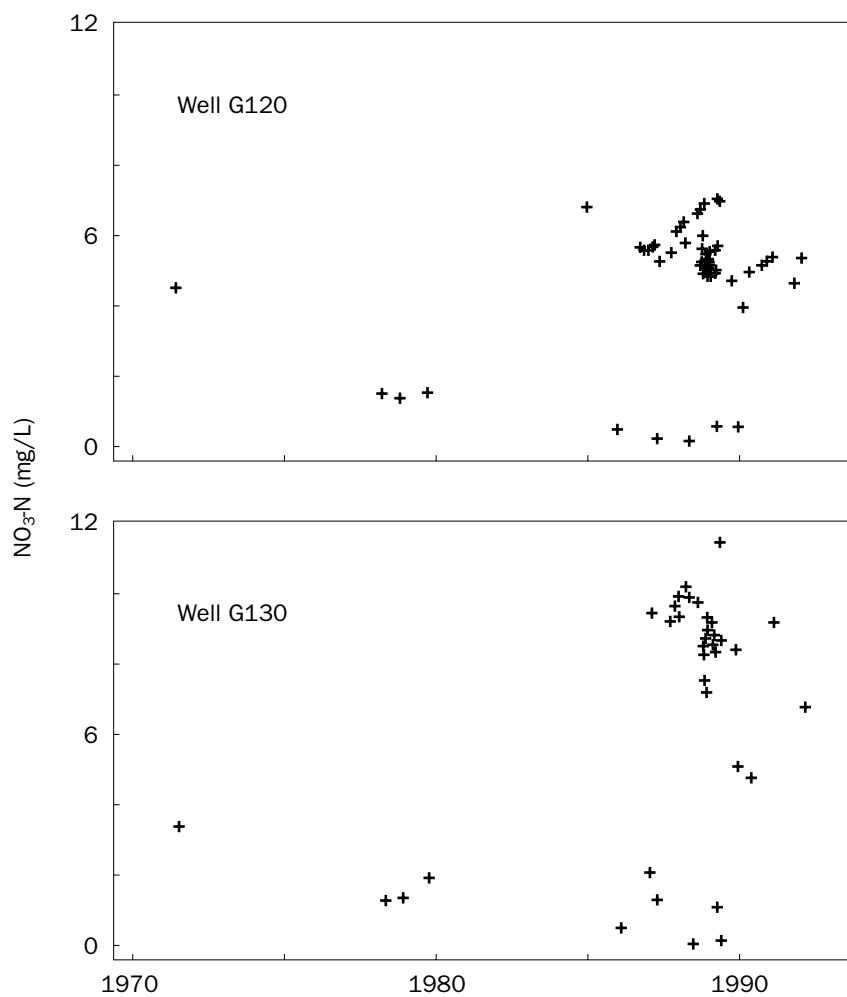
Sharma *et al.* (1992; 1993) found leachate nitrate concentrations were generally higher from sites on Bassendean sands, while groundwater nitrate concentrations are generally lower when compared to Spearwood sands. Appleyard (1995) reported nitrate concentrations in two urban areas of Perth to be in the range of 0.1–5.6 mg/L, and suggested denitrification accounted for the lower than expected concentrations within the aquifer. Barber *et al.* (1996) found denitrification within peaty soils, reduced nitrate impacts to groundwater from horticultural areas. The Waters and Rivers Commission (1997), found predicted groundwater nitrate concentrations versus actual nitrate concentrations of kennel subdivisions in priority 2 areas of Jandakot, to be lower than predicted due to denitrification within the aquifer.

As can be seen, denitrification or loss of nitrate from aquifer systems of urban areas of Perth is an important groundwater buffer to nitrate contamination from a range of sources. The lower than expected groundwater nitrate concentrations under Bassendean sands has been attributed to pH between 5.0 and 7.0, low redox potential's (< 300 mV) and anaerobic conditions. Therefore the identification of areas (not necessarily Bassendean sands) exhibiting any or all of these properties would provide important information that could be used in land-use zonings and management policy, designed to protect groundwater from nitrate contamination.

Other studies have examined the groundwater quality within and around the Gwelup Water Supply Scheme. Bestow (1981) found groundwater nitrate levels ranged from 0–1 mg/L, while ammonia levels ranged from 0–190 mg/L, beneath a land-fill site within the Gwelup water supply scheme. Maring *et al.* (1982) examined groundwater physical and chemical properties of the Gwelup Wellfield (Spearwood sands) and the Mirrabooka Wellfield (Bassendean sands). At the Gwelup site groundwater nitrate concentrations were up to 2.95 mg/L. The Mirrabooka site showed much lower nitrate concentrations, up to 0.12 mg/L. The Water Authority of Western Australia (1987) found groundwater nitrate concentrations to be between 1 mg/L and 29 mg/L at the Gwelup flow line (northern Perth Area).

Barber *et al.* (1993) reported that the Gwelup Underground Water Pollution Control Area showed elevated levels of nitrate in groundwater approaching the current drinking water standard of 11.3 mg/L. Barber *et al.* (1996), described land-use changes over 40 years within the Gwelup study area and found low nitrate levels under natural bushland, elevated nitrate concentrations in groundwater under areas which had a land use history of horticulture and market garden and septic tanks. Nitrate levels are low under parkland and new sewerred residential area. Nitrate concentration in groundwater below and downgradient unsewered areas increased in the last decades (Figure 17).

Figure 17
Increase in nitrate-N concentrations with time in two production bores downgradient of unsewered residential area (after Barber et al., 1996)



Groundwater nitrate levels at the Gnangara Mound and surrounding areas to the north of Perth appear to have generally higher nitrate concentrations than the Jandakot Mound to the south. The Dames & Moore (1986) report to the Water Authority of Western Australia found superficial formations of the Gnangara Mound to have relatively low concentrations of nitrate. While Pionke *et al.* (1990) found average nitrate concentration of 10.2 mg/L $\text{NO}_3\text{-N}$ in a deep groundwater site in Gnangara. Sharma *et al.* (1991) reported even higher groundwater nitrate levels beneath horticultural properties on the Gnangara Mound. As high as 80 mg/L at one horticultural property. Davidson (1995) however, found Gnangara Mound groundwater resources generally have nitrate concentrations less than 1 mg/L. There appears to be a pattern of increased groundwater nitrate levels within the Gnangara Mound, although justifying this pattern from the research presented here would be premature. Further insights may be given by examining the Water and Rivers Commission monitoring data.

Other urban areas of Perth have also been investigated for groundwater nitrate levels and showed trends towards greater groundwater nitrate levels in areas of new and old urban environments. Whelan and Parker (1981) found some unsewered areas north-east of Perth on Bassendean sands had groundwater nitrate concentrations as high as 70 mg/L. In 1983 the Australian Water Resources Council identified the Perth Basin as having nitrate rich groundwaters. In 1987 the Water Authority of Western Australia reported that groundwater samples taken from Kwinana, Coogee, South Fremantle, Bicton, Applecross, Midland and a few sites between Perth and Wanneroo had nitrate levels greater than 10 mg/L. It was also found that groundwater nitrate concentrations in excess of 1 mg/L occur exclusively within urban areas.

Gerritse *et al.* (1988) investigated the effect of urbanisation on the quality of groundwater along a north-south transect in Perth, Western Australia. Concentrations of all major ions in the superficial aquifer, were found however to increase in correlation with the level of urban development. The average groundwater nitrate concentrations reported for Nedlands/Dalkeith was 2.6 mg/L. Appleyard (1995) however reported nitrate concentrations in Whitfords and Nedlands to be in the range of 0.1–5.6 mg/L. It was also found that nitrate concentrations in groundwater under uncleared native vegetation (Barragoon) was half the concentration under two other urban areas (Whitfords and Nedlands). Generally the older the urban area in Perth the greater the general contamination of groundwater (Appleyard, 1995).

Various land uses have also seen increases in groundwater nitrate concentrations. A Department of Agriculture (1991) survey of groundwater nitrate concentrations from 40 bores in market gardens found that over half the bores had nitrate concentrations that exceeded the World Health Organisation Guideline of 10 mg/L $\text{NO}_3\text{-N}$ (Lantzke, 1995). Hirschberg (1991) found that extensive development in the Kwinana and Canning Vale areas has created new point sources of groundwater contamination. Davidson (1995) indicated that superficial aquifers of all urban areas of Perth especially surrounding coastal areas and the Swan River have nitrate concentrations higher than 1 mg/L, and that confined aquifers such as the Leederville formation all had nitrate concentrations less than 1 mg/L (Hopkins, 1996) expressed concern that as urban development increases, there is increased pressure to allow development to go ahead on groundwater mounds. Sharma *et al.* (1996) reported high leachate nitrate concentrations beneath urban lawns, due to the use of fertiliser. It has been estimated that the total area occupied by lawn requiring fertiliser in Perth is 13,500 ha and 67% of this is domestic lawn, while the rest is public parks and golf courses. By the year 2000 this would have risen to 16,000 ha with fertiliser applications for public and private lawns to be 100 kg/ha and 30 kg/ha respectively, resulting in the input of 1,600 tonnes of nitrogen; 20 to 30% of which would be taken up by plants. This leaves 70% to be leached beyond the root zone into groundwater and finally discharged to wetlands and waterways (Sharma *et al.*, 1996).

We investigated groundwater quality data for the last 10 years from 204 monitoring bores on the Jandakot mound and in the Jandakot UWPCA for possible trends in groundwater contamination by nitrate in relation to land use. Only 149 bores were considered near the UWPCA. Not many bores were sampled continuously or the data did not contain any nitrate concentrations. Numerous bores were not sampled at all, especially in the UWPCA. Figures 18 and 19 show the nitrate concentrations in the last eight years. There seems an increase in nitrate levels (>40 mg/L) to the southwest of the UWPCA. Overall the nitrate concentrations seem not to have changed much over the years. It seems that the spatial distribution of nitrate concentrations is controlled by when and where a bore was sampled.

The nitrate trend are plotted for six monitoring bores west of the UWPCA (Figures 20, 21 and 22). The nitrate concentrations fluctuate widely possibly due to different loading rates over time from the agricultural and animal farming land uses in the area. A local study is needed to establish the causes in relation to land use and farming practices.

From those investigations examined here, there appears to be a pattern of increased groundwater nitrate concentrations within the Gwelup and Gnangara UWPCA and a relationship between increased groundwater nitrate concentrations and further urban development. No conclusions could be drawn on increased groundwater nitrate concentrations within the Jandakot mound as nitrate concentrations appear to be consistently low, except for areas to the southwest of the UWPCA which need to be assessed in more detail in relation to land uses.

Future groundwater nitrate problems

As urban development within the Perth region increases so will the potential sources of nitrate that can leach to groundwaters. Increases in Perth's population will apply greater pressure to governments departments to allow urban development to encroach further on groundwater protection areas. Although future urban development is planned to be sewered, increases in urban lawns will mean increased application of fertilisers and therefore increased supply of nitrate to groundwater (Sharma *et al.*, 1996). The problems of increased groundwater nitrate contamination are not limited to future development. Barber *et al.* (1996), found a delay response of groundwater to increased nutrient input to be 1–7 years, while the full impact of previously unsewered areas may take 15–20 years. This means peaks in groundwater nitrate levels from previous, sometimes inappropriate land use has not yet been seen. Reverse partial tracking modelling has shown that in the Gwelup area production bores will be affected and need to be shut-down in the near future because of their downgradient location (Otto *et al.*, 1994).

These future threats to groundwater quality require that monitoring is carried out on a regular basis and that controls on land use restrictions within UWPCA's be strictly enforced.

Figure 18
Trends in nitrate concentration in groundwater for 1997 to 1994 on the Jandakot mound

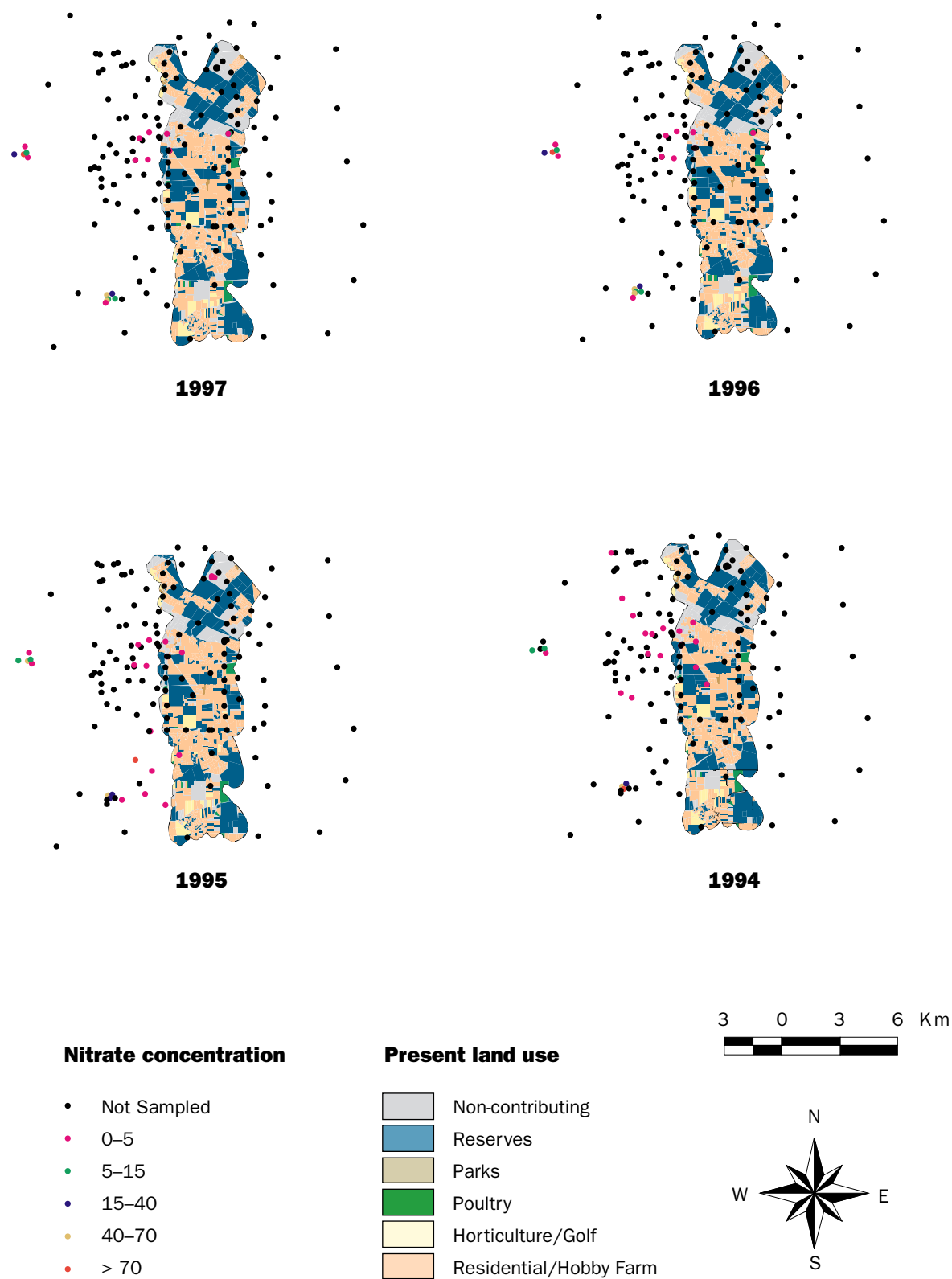
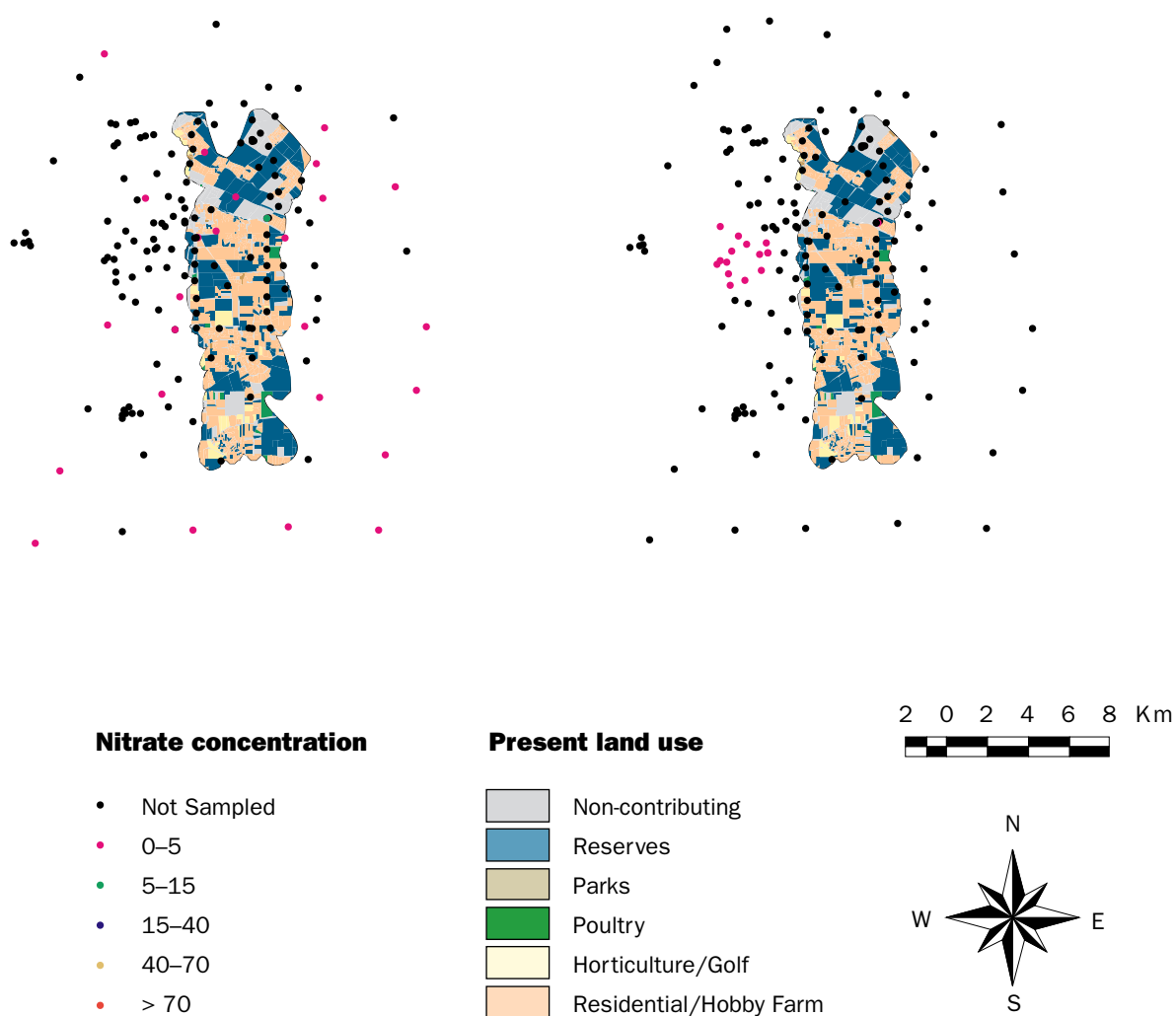


Figure 19
Trends in nitrate concentration in groundwater for 1993 to 1991 on the Jandakot mound



Recommendation for further work

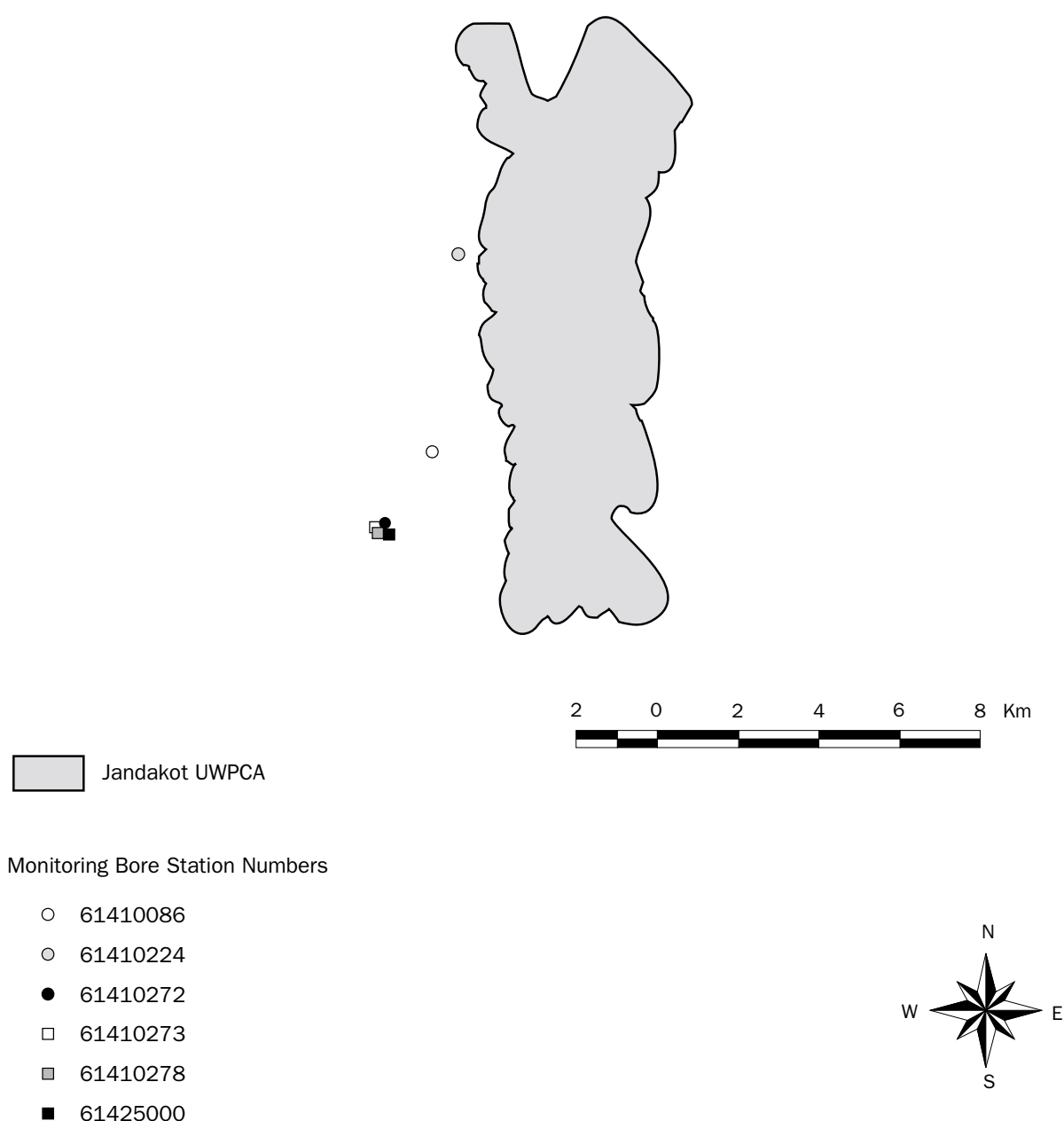
The rate and extent of denitrification in the soils and sediments, and the factors that affect it need to be investigated in order to provide actual data for the Perth urban areas and UWPCA's.

The leaching factor for nitrogen applied to different soils from various land uses need to be determined quantitatively.

The extent of leaking sewer pipes and the loss of nitrogen from sewers has never been investigated in the Perth metropolitan area.

Figure 20

Location of monitoring bores on the Jandakot mound* which had continuous groundwater quality data (see also Figures 21 and 22)



* Jandakot mound is 30 km south of Perth.

Figure 21

Concentrations of nitrate-N in groundwater west of the Jandakot UWPCA fluctuate over the years and months.

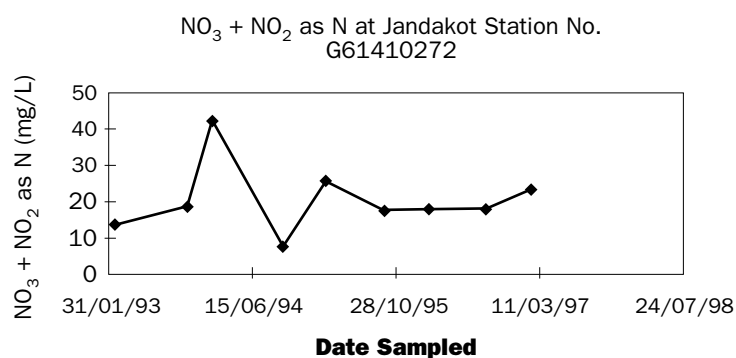
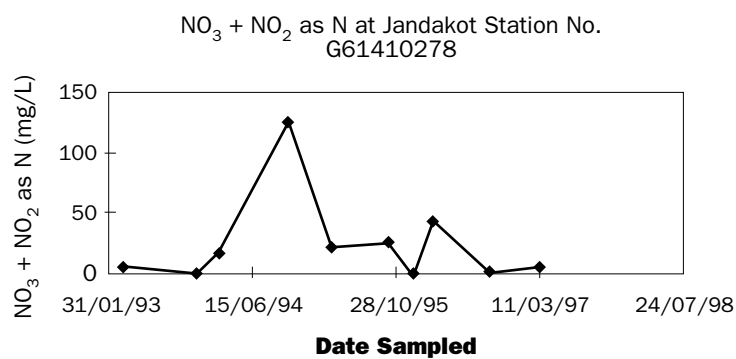
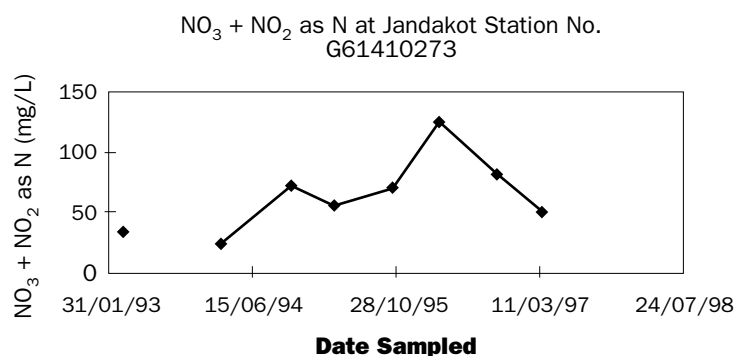
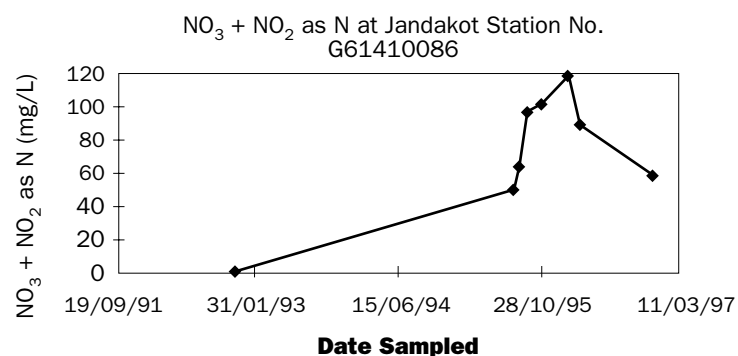
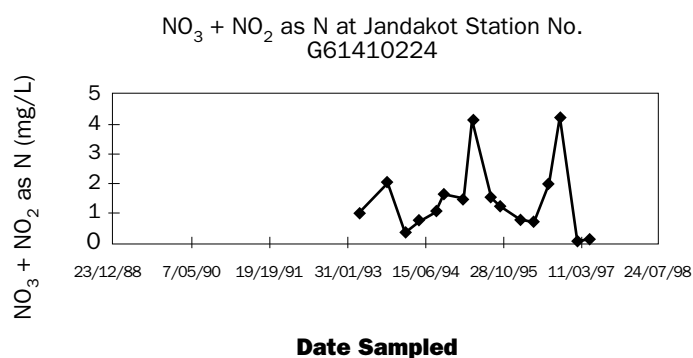
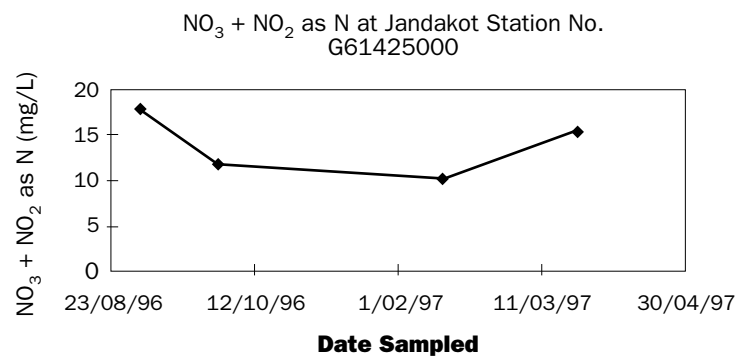


Figure 22

Concentrations of nitrate-N in groundwater west of the Jandakot UWPCA fluctuate over the years and months. Note a rapid increase in nitrate-N in bore G61410086.



Acknowledgments

The Water and Rivers Commission of Western Australia made the groundwater quality available. Mr Paul Carlile at the Centre for Groundwater Studies conducted the literature review, evaluated the groundwater quality database and analyzed the spatial distribution and time trends of groundwater contamination by nitrate using a geographical information system.

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Urban and rural residential— Wagga Wagga

Description of study area

Wagga Wagga is a city of over 56,000 population and forms a part of Murrumbidgee Valley.

Geology/hydrogeology

The city is underlain by slate and granite. There is an alluvial sequence up to 150 metres thick associated with the Murrumbidgee River. It comprises a highly permeable sandstone gravel sequence which underlies a shallow clay band units which may be up to 30 metres. Individual bores yield up to 200L/sec from the deeper unit and it considered to be a key aquifer.

Land use

The land use in the area varies. Wagga Wagga itself is an urban area. Dry cropping and grazing occurs in the surrounding area. An effluent treatment plant is located near the edge of the city and there is a landfill nearby.

There is also a feedlot within about 25 km of Wagga Wagga.

Extent of monitoring and nitrate concentrations

At the commencement of this study it was understood that nitrate was a concern in Wagga Wagga. There is a record of six monthly monitoring of Nitrate from monitoring of Town Water Supply Bores/ extraction points operated by Southern Riverina County Council, for Wagga Wagga and surrounding areas. The nitrate levels in the region are low, the highest being 2.3 mg/L nitrate as N, suggesting that the original assessment of elevated nitrate concentrations has not been confirmed.

The bores in the township of Wagga Wagga generally record Nitrate (as N) concentrations at least than 0.5 mg/L with occasional higher values up to 1.9 mg/L.

The Groundwater samples from bores in the bedrock aquifer at the Ladysmith Feedlot show no evidence of elevated nitrate. Similarly there are few records of nitrate above 0.5 mg/L in most of the regional areas around Wagga Wagga. However, areas of dry cropping in several areas have indicated nitrate up to 2.3 mg/L.

Discussion

There is no significant information on which to base major conclusions for the data in the at Wagga Wagga area. Due to lack of available data at this stage no trend analysis has been carried out. While there is no strong indication of elevated nitrate concentrations for the city water supply or the areas around Wagga Wagga, a number of points are noted:

- the low concentrations of nitrate at Wagga Wagga contrast with the elevated nitrate noted in other regional urban areas;
- the short record of nitrate recorded shows no apparent trends;

- the presence of cropping at the edges of the city does not appear to be introducing nitrate at concentrations high enough to elevate the levels above background levels;
- it is not clear if there is introduction of nitrate to the aquifer from typical urban sources such as urban storm water run-off, and that concentrations in the aquifer are being diluted by high groundwater throughflows in the gravel aquifer;
- the reported clay unit above the gravel aquifer is of variable thickness and its potential to act a protective cap to the main aquifer is not known;
- the significance of sewerage in minimising nitrate introduction to the groundwater may be significant; and
- dry cropping in the regions around Wagga Wagga show the highest concentrations of nitrate, even though they are low (<2.5 mg/L).

Intensive agriculture— horticulture and cropping—Bundaberg

Introduction

Intensive agriculture under tropical conditions using sugar cane as the major crop is practised in a number of areas along the Queensland coast. Significant nitrate contamination of groundwaters has been detected in a number of these groundwater systems, which is documented in the Bundaberg case study.

A number of different climatic and hydrogeological zones where there is intensive horticulture for other crops provide a different perspective on the Nitrate processes but there are fewer data. These are Shepparton East (VIC) where there is intensive horticulture for stone fruit and Rochedale (QLD) where there is a range of cash crops.

Location and hydrogeological setting

The aquifers in the Bundaberg area are a source of high quality potable water which is used for both urban supply and for irrigation, supplying around 80% of the water supply for Bundaberg. Groundwater is extracted from the Fairymead Beds which comprise interbedded sand, gravel and clay and some lignite and occurs at the base of the sequence in the Bundaberg Trough and from the overlying Elliot Formation consisting of sandstone, mudstone, and conglomerate. The Elliot Formation is the major aquifer for the water supply for Bundaberg. Minor quantities of water are also extracted from coastal dune sands.

In the south-eastern part of the Bundaberg area near Elliot Head there is also outcropping Basalt.

Soil types in the area vary widely, resulting in varying infiltration rates.

Land use and nitrogen source

Land use in the Bundaberg district has been dominated by sugar cane production for over 100 years, with over 90% of the agricultural land used for sugar cane production. Small areas of horticulture, including tomatoes and some corn and zucchini, have also developed in the area. The application of large amounts of nitrogenous fertiliser, provides a major source of nitrate for potential leakage to the underlying aquifers. Current application rates are assessed to be in the order of 140–180 kg N/ha (Keating *et al.*, 1997).

There has also been an expansion of urbanisation and rural residential development in the area with a resulting increase in potential sources of nitrogen for groundwater contamination.

Extent of known groundwater contamination identified in Bundaberg

Studies conducted

There has been ongoing sampling of groundwater from water supply and private bores over many years in the Bundaberg area. This has included some long-term regular monitoring but mostly single sampling events. These data are available in the QDNR data base.

Selected studies have been conducted over a number of years in the area to assist in evaluating the impacts of irrigation and fertiliser applications on sugar cane production, to identify the extent of nitrate contamination on the underlying groundwater system and its discharge points. A major study undertaken in 1993 by Queensland Department of Primary Industries, and CSIRO Division of Tropical Crops and Pastures (now Tropical Agriculture), was a snapshot survey of nitrate and biocide contamination which would form a baseline survey for evaluating future impacts.

This study included sampling and analysis of groundwater from 425 bores of which half were private bores. Most of the bores sampled were from the Elliot Formation. In addition this study documented trends from a number of representative urban water supply bores.

Further studies have been conducted in the area by CSIRO (Keating *et al.*, 1997 and Weier *et al.*, 1996) including detailed sampling of the saturated and unsaturated zones and detailed modelling to evaluated the impact of sugar cane management on both crop yield and on groundwater quality. This work is now essentially complete, although a low level of ongoing monitoring is being made by CSIRO Tropical Agriculture.

Results of monitoring

The results of various studies conducted at Bundaberg are presented in Table 15 (Keating *et al.*, 1996). The results from the various studies indicate that the groundwater in Bundaberg generally meets the World Health Organisational guideline concentration of 45 mg/L nitrate as Nitrate (10.0 mg/L Nitrate as N), although elevated concentrations above background levels are widespread. Keating *et al.* (1996b) report that of 425 bores sampled, 85% of bores in the area contain nitrate < 5mg/L Nitrate (as N), 10 % between 5 and 10mg/L and 5% exceeding the WHO guideline of 10mg/L nitrate as N.

Table 15
Information on nitrate concentrations from the Bundaberg area (from Keating et al., 1996)

Study	Time Period	No of Bores	Summary of Results	Comments	Source
1	1974 to 1990	4,782	2% > 100 mg NO ₃ /L 4% > 50 mg NO ₃ /L 7% > 13.5 mg NO ₃ /L	Samples not rigorously collected	QDNR Database
2	1974 to 1997	40	0% samples > 50 mg NO ₃ /L 68% samples > 13.5 mg NO ₃ /L	Data are averages of two to five samples over 3.5 years	Stickley (1980)
3	Oct–Nov 1993	425	4% samples > 50 mg NO ₃ /L 38% samples > 13.5 mg NO ₃ /L	First snapshot of both investigation and domestic bores	LWRRDC Study (BA Keating unpublished data)
4	Oct–Nov 1994	531	1% samples > 100 mg NO ₃ /L 4% samples > 50 mg NO ₃ /L 43% samples > 13.5 mg NO ₃ /L	Monitoring bore networks at 6 to 12 week intervals	LWRRDC Study (BA Keating unpublished data)
5	1993 to present	52	31% bores steady 15% bores rising 21% bores falling 33% bores fluctuating		

The main centres of high nitrate concentration coincide with the most permeable soils in the area and are in the highest recharge sites. However the results do not indicate that extensive plumes of highly contaminated groundwaters are migrating large distances away from the local nitrate sources.

In isolated areas, high concentrations also appear to be associated with urban development, particularly non-sewered areas (Weier *et al.*, 1996).

The depth of bores containing high nitrate concentrations is variable with most wells intersecting the shallow Elliot Formation. There is some evidence of shallow bores being associated with higher nitrate levels. There are also however a small number of high nitrate concentrations reported from deeper bores. A preliminary evaluation suggests that some of the deep bores receiving nitrate at depths could be perhaps associated with some fractured rocks especially the basalt.

The water quality trends for a number of key water supply bores which have been monitored from the early to mid 1970s although the quality of these data are thought to be highly variable. These show no consistent trends in Nitrate concentration over the nearly twenty year period (Figure 23). Two wells showed increasing nitrate, two were decreasing and the remaining two showed fluctuating results.

Detailed experimental sites

Detailed experimental sites in three areas where there is elevated nitrate in the groundwater have been investigated by CSIRO (Keating *et al.*, 1996). These studies involve establishing the major components of the nitrogen balance and using a modelling approach to simulate improved nitrogen management.

The monitoring of the bores around three experimental farms on a six week basis provides an indication of the response of the watertable and the nitrate concentration in the aquifer to rainfall. These results suggest that there is variability in the response of the nitrate concentrations to particular events such as rainfall (ie. recharge) events.

Multilevel bores at the experimental sites have been monitored for several years (Figure 24).

In particular there is an indication in several bores of slight increases in nitrate concentrations with time while there are no similar trends in most bores. Those showing slightly increasing trends are within the same cluster of bores and would appear to be in similar situations. There is a marked difference in response in some of the bores at different depths but at the same location. Likewise the response of the aquifer to recharge from rainfall is variable even at the same site.

Discussion

The data available at Bundaberg provides no conclusive evidence of increasing concentrations of nitrate in groundwater. However in some water supply wells a long-term increase has been detected. The data set however is small and the data collection is not systematic. Some of the monitoring is from production wells so that there is little control on the depth from which the samples are recovered. In addition the timing of sample collection is not consistent and does not relate to a consistent set of environmental conditions

There is a continuing source of nitrate in the region from continued application of fertiliser. Site specific studies by the CSIRO (Keating *et al.*, 1996b) indicate that there is leaching of nitrate in to the soil profile with the potential to migrate to the groundwater system.

The fluctuation of nitrate in both time and space suggests that there are varying conditions affecting the migration into the groundwater and the potential for preservation of nitrate.

Figure 23

Trends in nitrate concentration—water supply bores, Bundaberg

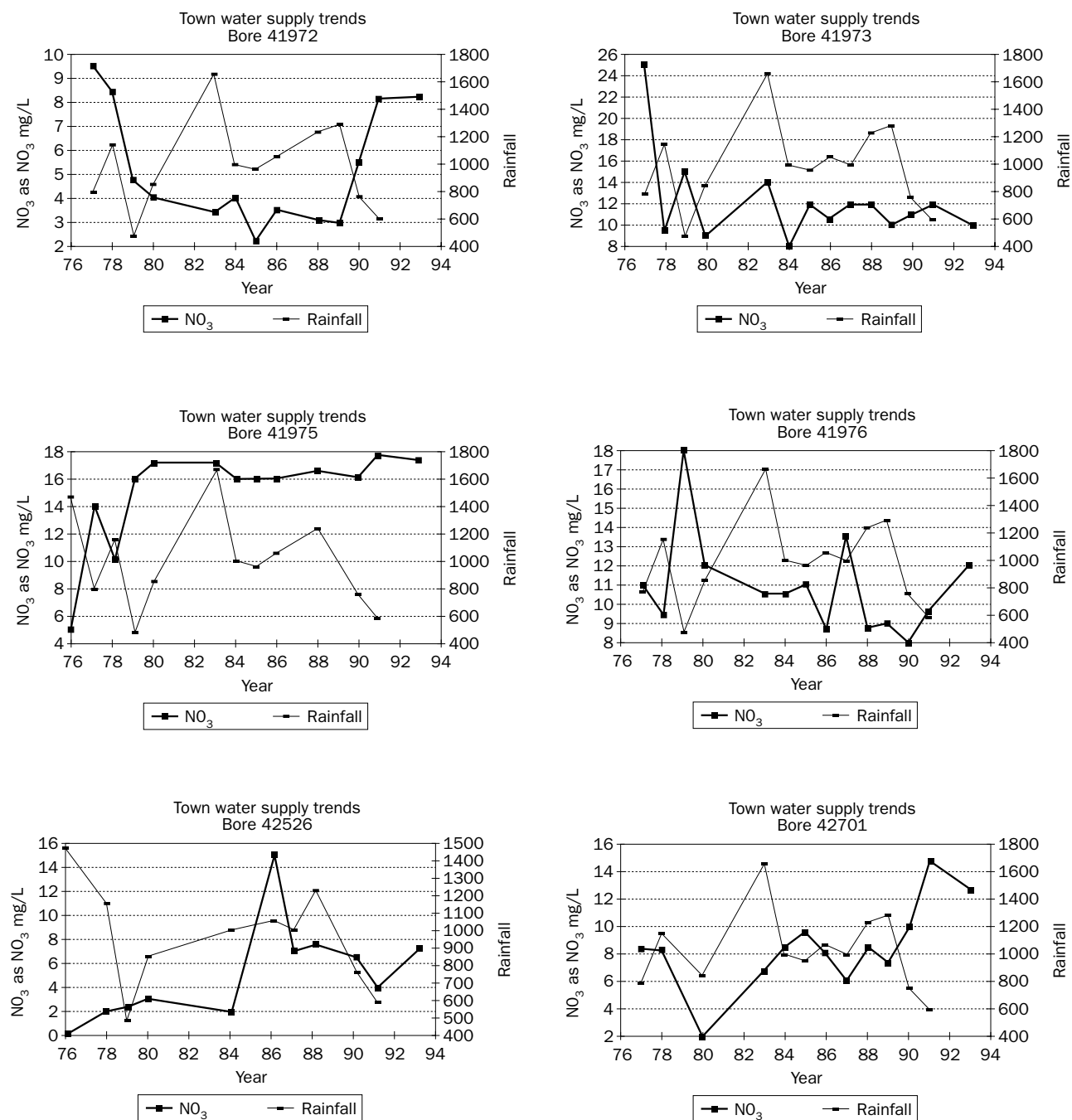
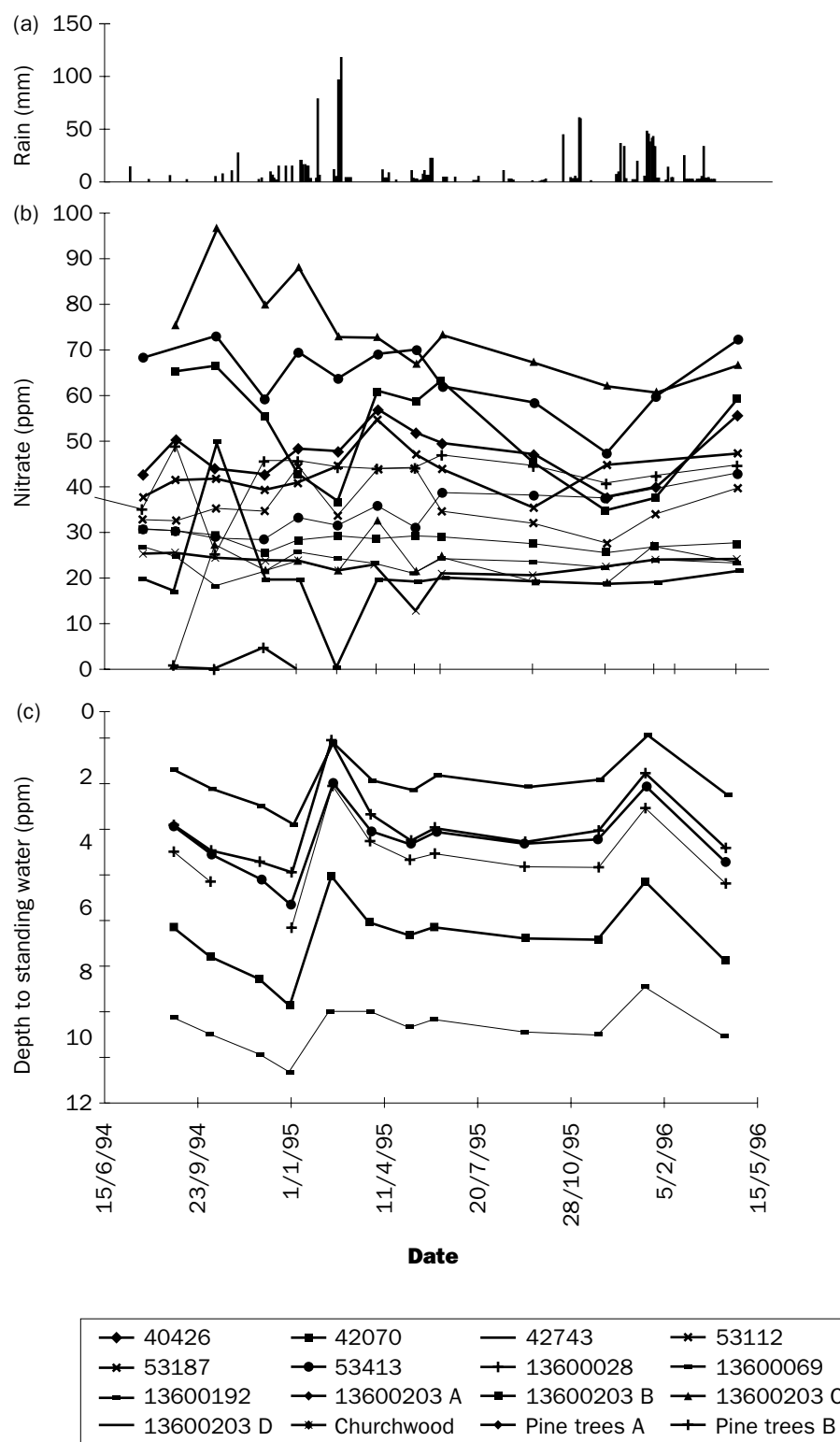


Figure 24

Rainfall (a), nitrate concentration (ppm as NO_3), and depth to standing water for 16 bores (after CSIRO, 1996)



There is the potential for denitrification in some parts of the profile which may be enhanced by the temperature and increased bacterial activity in the tropical soils. Likewise high rainfall events appear to have an impact on the nitrate concentration

There are major areas at which there is groundwater with nitrate at concentrations above the WHO drinking water guidelines. These are often house bores not associated with the irrigated agriculture and there is a significant political issue.

Conclusion

There is no conclusive evidence of major increases in nitrate concentrations in groundwater in the aquifers at Bundaberg over the last 20 years. However, there is extensive evidence that current nitrate levels have been raised above 'natural' levels through anthropogenic factors. In some areas, current levels exceed drinking water standards. Hence while agriculture and in some areas, urbanisation has clearly impacted on nitrate levels, rates of change over the last 20 years have not been so great that they can be reliably detected in view of the extensive variability and poor quality sampling conducted prior to the advent of the LWRRDC–CSIRO project in 1993.

Another factor that may be moderating continued rise in nitrate levels is the recirculation of groundwaters via irrigation use and other forms of loss, such as denitrification in the aquifer itself, which is poorly understood.

The available data suggests that in some areas there may be the potential for increases, particularly in the light of high recharge rates and continuing application of fertilisers at high loads.

On some sites, high levels of nitrate have been found in the sub-soils and a possibility exists that this may ultimately find its way to the aquifersystem.

The LWRRDC–CSIRO Study in Bundaberg has stimulated a number of followup activities. Some deal with reducing nitrate losses from intensive agriculture (management of trickle irrigation, finetuning fertilises rates via crop testing). Another SRDC project is undertaking a major survey of groundwaters associated with all the sugar growing areas in Queensland and NSW (KL Weier per comm.) Included in this work is continuation of the quality monitoring of bores in the Bundaberg area. This monitoring will be important in clarifying current trends.

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Mixed agricultural land use— Peel Valley, NSW

Description of study area

The study area is situated around the rural township of Tamworth in northern NSW.

Geology/hydrogeology

The geology of the surrounding rocks is complex. In general it consists of metasediments covered with a thin colluvial layer, and granite with a thick residual regolith in some areas. There is residual tertiary basalt capping the catchment divides in the east. Alluvial flats occur along the Peel River. They are usually less than 1.5 km in width, but between Tamworth and Attunga, attain a maximum width of about 3 km.

The watertable is at a depth of around –2–3 m in the alluvium, but appears to vary from around 5–12 m in the fractured metasediments and around 5 m in the granite.

The main source of groundwater in Peel Valley is unconsolidated alluvium of the Peel River and its tributaries. The alluvial system comprises shallow (less than 15 m) clays, silts, sands and gravels. Individual bore yields in the alluvium are less than 15 L/s.

The weathered mantle is up to 25 m thick on the fractured granite in the north-east of the catchment. Groundwater yields of up to 2–3 L/s occur in the deeper drainage lines in the granitic terrain.

Small groundwater supplies (generally less than 1 L/s) are also obtained from fractured rock in the catchment.

Land use

There is a range of land uses in the Peel Valley. The most predominant is grazing, dry cropping, with other important industries irrigated cropping, intensive animal husbandry (mostly piggeries, poultry farms) and rural residential.

The main industry on the alluvial flats is irrigated lucerne cropping, with no agri-industry on the river flats. Manufacture of fertiliser from poultry waste has been undertaken in the catchment on the fractured rock part of the catchment. Recently there has been a change in the management practices in this industry with improved sealing of surfaces and on-site drainage.

Extent of monitoring

In 1992, the now Department of Land and Water Conservation (then the Department of Water Resources) carried out groundwater quality survey in the Peel Valley. This was a single snapshot survey and only provided a spatial distribution of nitrate contamination.

Nitrate contamination

60 samples were analysed for nitrate in the field. 50% of these samples showed nitrate levels above 10 mg/l of nitrate as N; 47% between 5–10 mg/l of NO₃ as N and 3% between 5–10 mg/l nitrate as N. No other regular monitoring of nitrate is being carried out. Distribution of nitrate values in mg/l as N is given in Figure 25.

The overall pattern of nitrate contamination suggests that there are high concentrations of nitrate (>10 mg/L and including very high values of 52–170 mg/L) associated with grazing in the area, particularly in the granite country around the Cockburn River. Elsewhere (eg. Attunga Creek headwaters), grazing has had minimal impact on the groundwater with nitrate concentrations averaging less than 2 mg/L.

Both piggeries and poultry activities have had significant impacts on the groundwater. Nitrate concentrations associated with Poultry production is typically greater than 10 mg/L with one value up to 42 mg/L. Piggeries appear to mostly result in higher concentrations with a range of values from 24–52 mg/L.

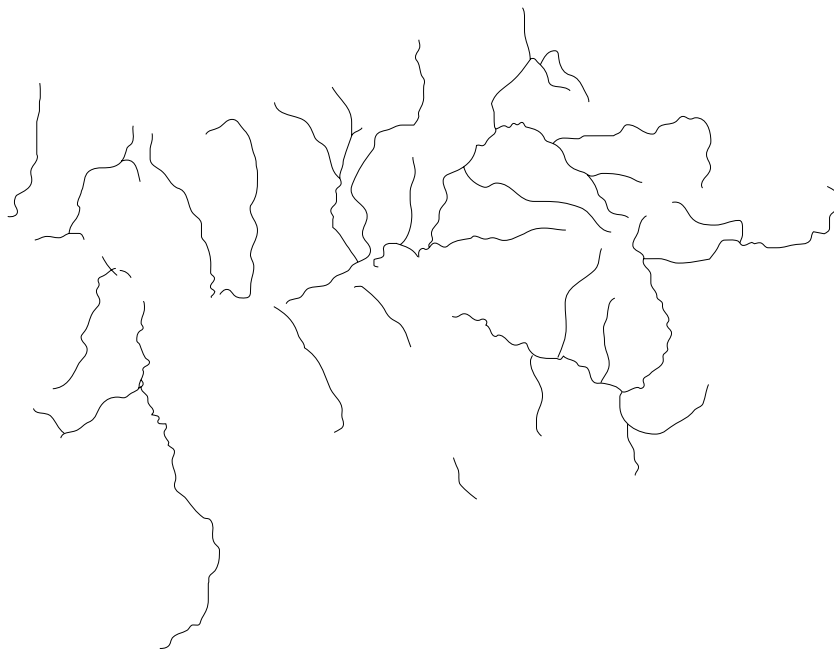
Irrigated horticulture, mostly lucerne, is often low (<2 mg/L) although there is one occurrence of 58mg/L a few kilometres upstream

Trends and future directions of nitrate contamination

As there is only one snapshot of data for the region, it is not possible to evaluate the trends or future directions in nitrate contamination in the Peel Catchment. However the distribution of high concentrations of nitrate across a wide range of industries suggest continuing potential for nitrate contamination.

It is noteworthy that the effect of grazing varies, although there is no apparent reason from this one set of data. Further evaluation of the land use history will need to be conducted to attempt to establish the possible impacts of different land use

There is understood to have been a change in management practice in the poultry industry in the area. This is likely to result in a reduction in nitrate loads to groundwater in the area.



Country town, rural cropping— Narromine, NSW

Description of study area

Narromine township lies in the Macquarie Valley and is about 40 km west of Dubbo. This area was selected because nitrate levels have been measured by Narromine Shire Council over the last two decades.

Geology/hydrogeology

The study area is underlain by approximately 100 metre thick alluvial deposits associated with Macquarie River. It comprises distinct clay materials which are interbedded with sand/gravel units up to 20 metres thick. The alluvium overlies an Ordovician/ Silurian sequence of slate, quartzite, shale and sandstone.

The main aquifers are sand/gravel units which occur at depths generally greater than 40 metres. The transmissivity of these aquifers range from 170 m²/day to 380 m²/day.

The regional groundwater movement in this area is from north-east to south-west.

Land use

This case study considers land use in a regional country town. Irrigation cropping occurs outside the town.

Extent of monitoring

Narromine Shire Council has been regularly monitoring nitrate levels in town water supply bores since the early eighties. The location of the bores in relation to the town is shown in Figure 26.

Nitrate levels in most of the town water supply bores fluctuate over the period showing no set pattern in change of levels. Nitrate levels remained below 10 mg/L of nitrate as N. However bore 7 which is located on the southwest side of the town in a downgradient direction has consistently recorded nitrate concentrations up to a maximum of 8.9 mg/L. Other bores even those upgradient of the town typically range from around 2 to 5 mg/L nitrate. Occasionally the nitrate concentration reaches higher levels of above 7 mg/L.

Discussion

The source of the nitrate in the groundwater at Narromine is not clear although there are considered to be numerous potential sources

Harwood (1986) identified eleven potential sources of nitrate. These are:

1. point source;
2. agricultural chemical applied to the ground surface;
3. sewage effluent;
4. leaking sewer mains;
5. stormwater drainage;

6. cattle yards;
7. garbage tip;
8. cemetery;
9. agricultural chemicals discharging into Macquarie River;
10. natural occurrence of nitrate in groundwater system; and
11. anhydrous ammonia plant within town limits.

Australian Nuclear Science and Technology Organisation (ANSTO) sampled the most contaminated bore and soil samples from the surrounding area were analysed for nitrogen isotopes in 1992. They concluded that the most of the nitrate contamination is mostly attributed to the animal waste.

Keshwan *et al.* (1994) compiled a report on Groundwater Quality Monitoring for Narromine Town Water Supply Bores.

Jewell (1996) concluded that pumping from town water supply boreholes generate an extensive cone of depression. The capture zone of the town water supply extends beneath the entire urban area. He concluded that the distribution of nitrate concentration suggest that leachate from municipal landfill may make a contribution to the overall problem but it is not a primary source. Rather he further suggested that any of the sources identified by Harwood (1986) may be contributing to the problem. Slow leakage through the upper clay sequence may delay impact for many years. There is no data or any suggestion to estimate this period.

It is also reported that during the pumping test (24 pumping period) concentration of nitrate in the discharge from TWS 7 increased by 0.5 mg/l suggesting significant local concentration. This may imply the presence of a store of nitrate within the soil zone which may provide an ongoing source of contaminant to the groundwater under suitable conditions.

A conceptual model of nitrate leaching based on nitrogen species movement compiled from all available information is given in Figure 27.

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Figure 26
Plan of the Narromine Town Area

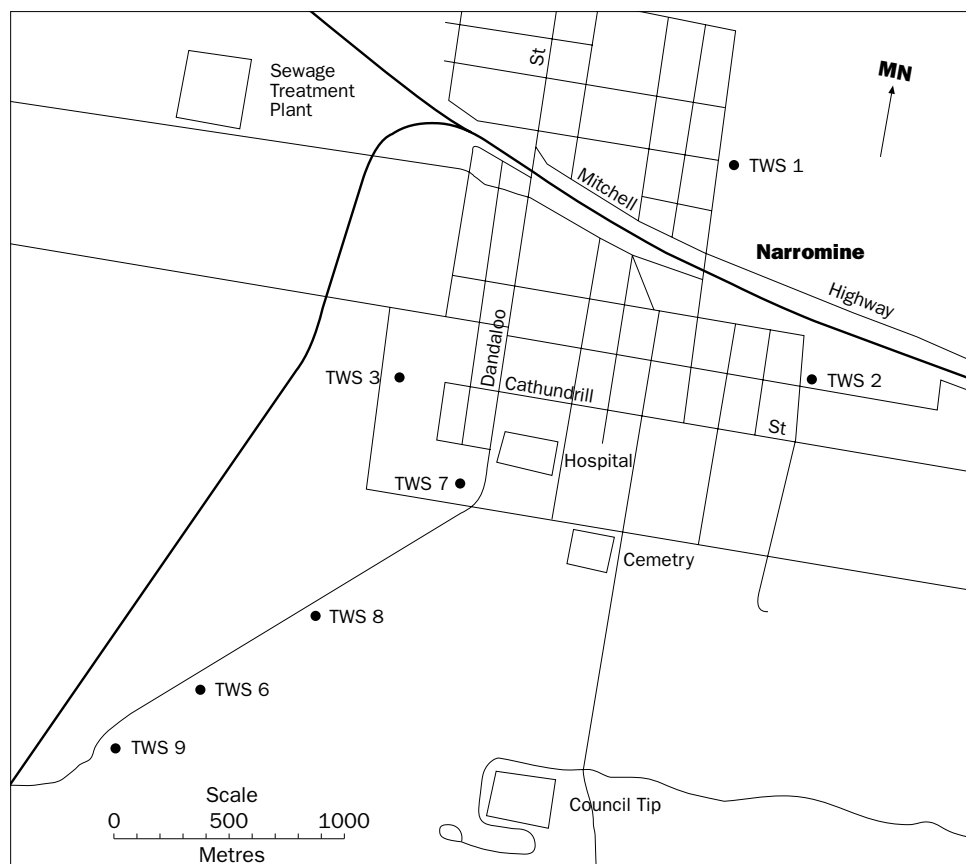


Figure 27
Conceptual model based on nitrogen species movement at Narromine

