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Agricultural Impacts on Groundwater Quality

Proceedings of an international workshop held in Kota Bharu, Kelantan, Malaysia, 24–27 October 1994

Editors: B.Y. Aminuddin, M.L. Sharma and I.R. Willett

Australian Centre for International Agricultural Research

Contents

Workshop Summary

THESE PROCEEDINGS arose from a meeting held in Kota Bharu, Malaysia, to review the findings of an ACIAR-funded project entitled 'Environmental impact of agricultural practices on water resources of the Kelantan Plain, Malaysia'. Additional information on agricultural practices on groundwater quality in neighbouring countries (southern Thailand, Indonesia, and the Philippines) was also presented in an attempt to get a more regional perspective on the impact of agriculture on groundwater quality.

Groundwater contamination has been recorded in detail from many industrialised countries of temperate regions. This has been linked to intensive agriculture in these areas, and associated applications of pesticides and high rates of nitrogenous fertiliser, as well as cultivation practices which favour leaching of nitrate below the rooting zone of plants.

There has been much less research on the effects of agriculture on groundwater quality in tropical regions. Nevertheless, tropical agriculture includes some highly intensive forms of farming with significant inputs of fertilisers and pesticides, and in many areas rainfall which results in leaching for at least some part of the year. The main examples are high yielding rice production in irrigated areas and the production of high value cash crops such as vegetables and tobacco. The ACIAR project in the Kelantan Plain is the first detailed and systematic study of groundwater contamination in the tropics.

The Kelantan Plain, located in the extreme northeast of Peninsula Malaysia, is an extensive (0.2 million ha) river flood plain with two intensive agrosystems — rice on clay soils and tobacco on sandy soils. The groundwater resources of the plain are the most extensive in Malaysia and are the source of drinking water for the majority of the rural population.

The research reported in several papers arose out of detailed studies of the Plain. A broadscale survey was conducted to determine if the groundwater was contaminated, and if so, how widespread it was. These studies were followed up with more intensive research at sites representative of the rice and tobacco producing areas.

Ahmad et al. presented results of the analyses of groundwaters from existing drinking water wells and piezometers for inorganic parameters (pH, total dissolved solids, NH_4-N, NO_3-N, P, K, Mg, Fe, Cl, electrical conductivity and redox potential). In the tobacco growing areas on sandy soils the levels for NO_3-N were found to exceed those for drinking water quality (10 mg/L) in 15% of the samples from drinking water wells. Most samples exceeded the limits for Fe and had pH values lower than recommended for drinking water. These results and those for redox potential showed that the groundwater is a reduced environment. Nitrate concentration in groundwater was generally low in samples drawn from reduced groundwater and higher in oxidised groundwaters. In the case of Fe, at least, the high concentrations reflect natural conditions rather than contamination by agriculture. However, there was evidence that NH_4-N and NO_3-N concentrations were related to surface applications of nitrogenous fertilisers.

In groundwater samples from rice growing areas Fe, and NH_4-N were at concentrations greater than those recommended for drinking water. These results also reflect the reduced conditions of the groundwater. Concentrations of dissolved nitrogen were lower under rice growing than tobacco growing, and indicated the tobacco areas were at greater risk of groundwater contamination by NH_4-N and NO_3-N. Point sources of nitrogen, particularly toilets and poultry sheds, contributed to NH_4-N and NO_3-N in groundwater.

The groundwater samples were also analysed for a range of pesticides. It was found that endosulphan (as its beta-isomer and its oxidation product) was ubiquitous in that it was found in most piezometer, wells, surface drain, irrigation canal, and river water samples. It was more widespread in the rice growing areas than the tobacco growing areas because it was more heavily used (until recently) on rice crops. Lindane residues were also commonly detected in both cropping systems, but at low concentrations. Traces of paraquat were detected. This may be due to its transport through preferential flow pathways as it is strongly sorbed by soils. Pesticides in current use were not detected in groundwater below tobacco. Some pesticides (carbofuran, aldrin, dieldrin, DDT) now banned from use were detected at low concentrations. Modelling work showed that pesticides move slowly through the soils (lindane and dieldrin take 5.5 and 40 years to travel 1 m, respectively) and transport of pesticides applied in the past can be expected to continue for many years (approximately 30 to 80) after their application to tobacco grown on sandy soils.

Analyses of total coliform bacteria (as colony forming units, CFU) showed there was widespread contamination of groundwater and drinking well water. There were large variations in the CFU counts both in time and space, but it was clear that coliform bacteria increased abruptly after each monsoon season. The sources of the coliform bacteria have not been determined, but would include septic tanks as well as grazing animals and poultry.

Techniques for evaluation and prediction of agrochemical pollution were also described. The methods of sampling the soil water by extraction from soil samples, ceramic cups and by lysimeters were compared. The lysimeter results showed less variation than the other methods. The application of models to describe water movement, nitrogen transformations and pesticide behaviour in soils have shown promise in allowing prediction of nitrate and pesticide accessions to groundwater. Successful application of a model to nitrate transport in intensive agriculture on sandy soils of Western Australia was described, and modifications for its use in the Kelantan Plain were outlined.

In comparison with the Kelantan Plain there appears to be very little information available in neighbouring countries. There was evidence of pesticide contamination of surface and groundwaters of southern Thailand. The Rattaphum watershed was presented as an example, and it has some features in common with the Kelantan Plain in that intensive vegetable production occurs with heavy fertiliser and pesticide applications in an area where the rural population relies on groundwater for domestic consumption. There have been some analyses of pesticide residues in water samples, and BHC, DDD, aldrin and endrin have been detected. However, there have been no systematic studies to establish the degree of contamination.

Wetselaar et al. described the distribution of nitrate in drinking water wells in villages in East Java, Indonesia. High concentrations of nitrate were found in many wells showing that millions of people are drinking well water with nitrate concentrations far in excess of those recommended for drinking. The source of the nitrate was not from rice production in fields surrounding the villages, but was associated primarily with the populated parts of villages. The most likely source of nitrate was septic tanks which had plumes of high concentrations of nitrate. Drinking water wells within nitrate plumes will be contaminated. Solutions for these problems include appropriate placement of wells in relation to septic tanks, piping and disposal of human wastes, and piping and treatment of drinking water.

There is much less information available from elsewhere in Indonesia. Rice production has been intensifying with increasing applications of fertilisers and pesticides. Some analyses of surface waters, which are used for drinking water in rural areas, have shown levels of nitrate above those recommended for drinking water, but there appears to be no data for groundwater. Intensive vegetable production in highland areas may result in accession of agrochemicals to the groundwater. There was also little information from the Philippines. On the basis of the quantities of agrochemicals applied and the light texture of the soils, sugar cane producing areas were identified as those that may have problems of groundwater contamination. This may well be the case in other countries also.

The discussion periods at the meeting recognised the importance of the Kelantan project as an important and probably the first tropical case study of agricultural pollution. Nevertheless, there are many other agricultural systems in Southeast Asia which may also be at risk of contamination of the groundwater. It is obviously not possible for each case to be subjected to the detailed work carried out at Kelantan, although one aim of the ACIAR project was to enable MARDI to carry out research in other areas. It was concluded that models of water and agrochemical transport have shown sufficient promise that they could be used for evaluation of contamination problems in other areas without the need for detailed (and technically difficult) field monitoring for long periods. However, a minimum set of input data for such models needs to be defined. In addition, the models require adapting to make them easier to use.

The discussion revealed the lack of coordination of efforts between authorities responsible for water supplies, health and agriculture. In some cases it appeared that there were conflicts of interest. Better information sharing and the recognition of the common goals of resource protection and provision of safe drinking water are needed. It was also concluded that the research effort should move on from only identifying the severity and extent of agricultural pollution toward devising practical management techniques to reduce the problems. Such effort could also be linked to modelling work.

In general terms it was concluded that priority should be given to systems with intensive production of vegetables and sugar cane rather than rice-based systems. This is related to the heavy and frequent applications of agrochemicals in combination with light textured soils that have limited capacity to retain applied chemicals against leaching, and periods of rainfall/irrigation in excess of evapotranspiration. Although rice production in some areas involves heavy applications of agrochemicals, the clayey soils normally used have a high capacity to retain agrochemicals where they will be degraded by chemical and microbial activity. In addition, the reducing conditions usually found in rice growing areas decrease nitrate accessions to the groundwater as they favour denitrification of nitrate and its loss from the system in gaseous forms.

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Environmental Impact of Agricultural Inorganic Pollution on Groundwater Resources of the Kelantan Plain, Malaysia

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Abstract

Many farmers of the Kelantan Plain are dependent on the shallow groundwater as the source of domestic and agricultural water supply. The main crops grown are rice and tobacco. Relatively high levels of fertilisers are used in tobacco and rice agrosystems it is very likely that these inputs could contaminate the shallow groundwater. This paper assessed the groundwater pollution by conducting a broadscale survey which measured inorganic fertiliser inputs in relation to landuse, soil type and agronomic practices. Thirty sites were sampled on the sandy tobacco area, and 30 sites were sampled in irrigated rice areas. At each site in both agrosystems, water samples were collected monthly from field piezometers and household open wells. In addition, a more detailed site-specific study was conducted to identify the factors and processes that influence the levels of the pollutants in the groundwater. Sixteen piezometers were installed on a grid pattern and sampled on a weekly basis. Data for the tobacco agrosystem indicated that on average, the level of NH_4-N in the groundwater was high (about 1 mg/L) in June 1992. Drinking well data indicated that nitrate could get into groundwater very quickly during the months of fertiliser application (March and April). The nitrate levels increased again during August until October with corresponding decreases in NH_4 levels. The trends of the levels of NH_4, P and K were substantially influenced by the depth of the water table and redox potential. In the rice agrosystem the average levels of NH_4 in the piezometers and drinking wells were generally below the permissible limit except in the month of July 1992, which indicates a direct influence of fertilisation on the groundwater. The specific site study indicated generally that the water table strongly influenced the redox potentials, which in turn controlled the levels of ammonium, nitrate, phosphate and potassium. The fluctuation of nitrate levels in the groundwater at several grid points, corresponded well with the water table depth and redox potential variations.

In AGRICULTURAL systems, water and fertiliser are important for successful crops. Due to the lack of knowledge and experience, farmers often supply these inputs in excess, which can cause impairment of the environment. Excess nutrients, especially N and P, cause eutrophication of surface waters and are responsible for accelerated growth of algae and other undesirable aquatic plants. These plants can cause blockage of waterways and can trigger flash floods after heavy rainfall. Furthermore, N in the form of nitrate can be leached into groundwater. Contamination of groundwater by nitrate is potentially an important environmental problem, in relation to the quality of water for human (Dillon et al. 1989; Hornsby 1990 and Sharma et al. 1991) and animal (Blood et al. 1983) consumption.

The groundwater resource of the Kelantan Plain is the most extensive in Malaysia, and the source of domestic and agricultural water supply for many farmers (ENEX 1977). The agriculture in the Kelantan Plain is mainly rice and tobacco, depending on the soil type. Tobacco is grown on sandy soils while rice is grown on clay soils. Relatively high levels of fertilisers are being used. The current yearly rates for NPK are 160, 80, 80 kg/ha for rice and 60, 50, 135 kg/ha for tobacco, respectively. The impact of these agricultural practices on the environment, especially on groundwater resources, has not been investigated.

In the tobacco agrosystem, the groundwater is shallow (<2 m) and during the wet season the water table

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is close to the surface (Mohammud et al. 1990). The vertical transport of pollutants through the sandy soil profile occurs not only from excess precipitation, but also from irrigation water. On farms with very small holdings, the wells are very close to the field, and are usually only a few metres from toilets which may discharge into the shallow groundwater. Sandy soils have low amounts of clay or organic matter to lessen the transport of pollutants.

It is highly probable that groundwater is being polluted by nitrate. It has also been found that leachate draining below the root zone, under heavy fertiliser conditions, contained high concentrations of nitrate and P (Sharma 1991).

In the rice growing areas nearly all the farm houses are dependent on shallow (<2 m) well water. Usually wells are located very close to rice fields. Despite the clayey texture of the soil, there is evidence of a link between irrigation water channels in the rice fields and water in domestic wells. Clayey soils used for rice may be able to retain P to a large extent. An additional source of contamination is private toilet systems which are usually installed close to the household wells.

In the Kelantan Plain, there is evidence of eutrophication, as algae and aquatic plants have proliferated in surface waters. It is highly probable that the shallow groundwater in the region has been contaminated, contributing to a health risk, since groundwater in this region is widely used by farmers and other rural population for domestic consumption.

The objectives of this paper are to assess the groundwater pollution by inorganic agricultural inputs in relation to landuse, soil type and agronomic practices, and if possible to identify the factors and processes that influence the levels of the pollutants in the groundwater by analysing in more detail data from specific sites.

Materials and Methods

Sampling Sites

A broadscale survey in rice and tobacco areas was conducted to monitor the possibility of groundwater contamination due to agricultural inputs. A total of 58 sampling sites were selected throughout the study area (Fig. 1). Twenty-eight of these sites were located on the coastal bris (sandy) soil complex, and two sites were located in the tobacco/rice cropping system on inland riverine alluvium soils. Similarly, 28 sampling sites were selected in irrigated rice areas with sampling sites chosen in areas where shallow groundwater is used for household consumption. In both agrosystems, each site was characterised in terms of location, landuse, soil type, crop management, fertiliser applications, and household details.

At each site in both agrosystems, water samples were collected mainly from field piezometers and open household wells using a peristaltic pump with an in-line filtration system.

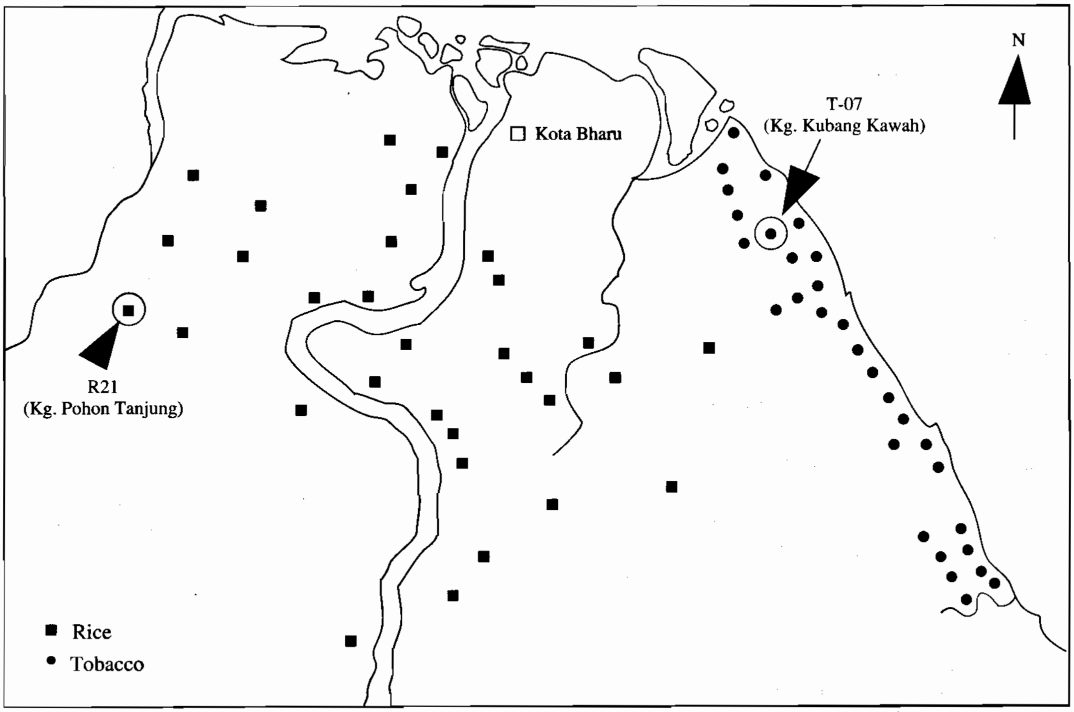

Figure 1. Location of sampling site in tobacco and rice areas.

Sampling Time

Sampling was categorised into intensive and extensive. Both organic and inorganic analyses were done for intensive sampling while only inorganic analyses were carried out for extensive sampling. Intensive sampling for tobacco was done in March, May and December and extensive sampling in April, June, August, and October. In the rice area, intensive sampling was done in January, March, July and September, and extensive sampling in May and November. Groundwater was sampled from within 1 m of the water table.

Inorganic Analysis

Two 250 mL water samples were collected. One sample was acidified to pH 2 with HCl to inhibit the oxidation and nitrification process, and was analysed for NH_4-N, NO_3-N, P and Fe. The other sample was not acidified and analysed for K, Ca, Mg, and Cl. NH_4-N and Cl were measured using an ion-selective electrode. NO_3-N was analysed using an auto-analyser. Inductively Coupled Plasma (ICP) spectrometry was used to determine P, K, Ca, and Mg. Electrical conductivity (EC), pH, redox potential (Eh), total dissolved solids (TDS), temperature and depth of the water table were recorded in the field using a portable conductivity meter, pH meter, redox potentiometer, TDS meter, thermometer and water level recorder.

Specific Site Studies

Following the broadscale surveys, detailed studies are being conducted at two representative sites. These are site T-07 (Kg. Kubang Kawah) in the tobacco area and site R-21 (Kg. Pohon Tunjung) in the rice area (Fig. 1). Only limited data from T-07 are reported in this paper. See Sharma et al. and Ayob et al. (these proceedings) for more information.

Site T-07 is located in Kg. Kubang Kawah near the eastern coast (Fig. 1). Its distance from both sea and river is about 0.8 km. The site elevation is about 3 m above sea level. The soil in the study area is characterised by loose, fine sand texture (<10% clay) and is well-drained. The soil is developed on sandy beach ridges, with some riverine influence. For convenience of cultivation and other farming operations, the farm in the site T-07 has been divided by farmers into uneven sized plots (Fig. 2). An earthen drain has been built to enable groundwater discharge to minimise the impact of flooding.

Sixteen piezometers were installed to a depth of 2 m on a 36 m grid pattern. Two additional piezometers (P1, P2) were installed near the household. With the inclusion of a drinking well, a total of 19 water samples were collected on a weekly basis over a tobacco growing period. At each sampling, water table, temperature, pH, Eh, TDS and EC were measured. The methods of sample collection and laboratory analysis were the same used in the broadscale survey.

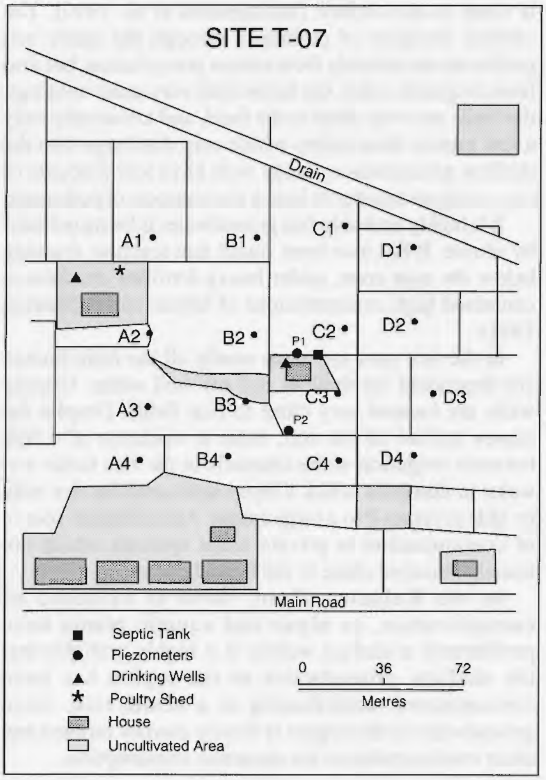

Figure 2. Map showing the location of piezometer in relation to drinking wells and other identifiable features at T-07.

Drinking Water Standards

A number of standards have been used to compare water quality and permissible levels of chemicals in groundwater. Most of the industrialised countries have developed some form of enforceable drinking water standards to ensure public health. Data in Appendix 1 compares the inorganic standards/guidelines developed for Australia, Canada, EEC, Malaysia, USA and WHO. The Malaysian drinking water standard was used as a reference for the minimum acceptable level. Where Malaysian standards were not available, EEC standards for K (10 mg/L), EC (400 S/cm) and Mg (30 mg/L), and WHO standards for TDS (1000 mg/L) were used.

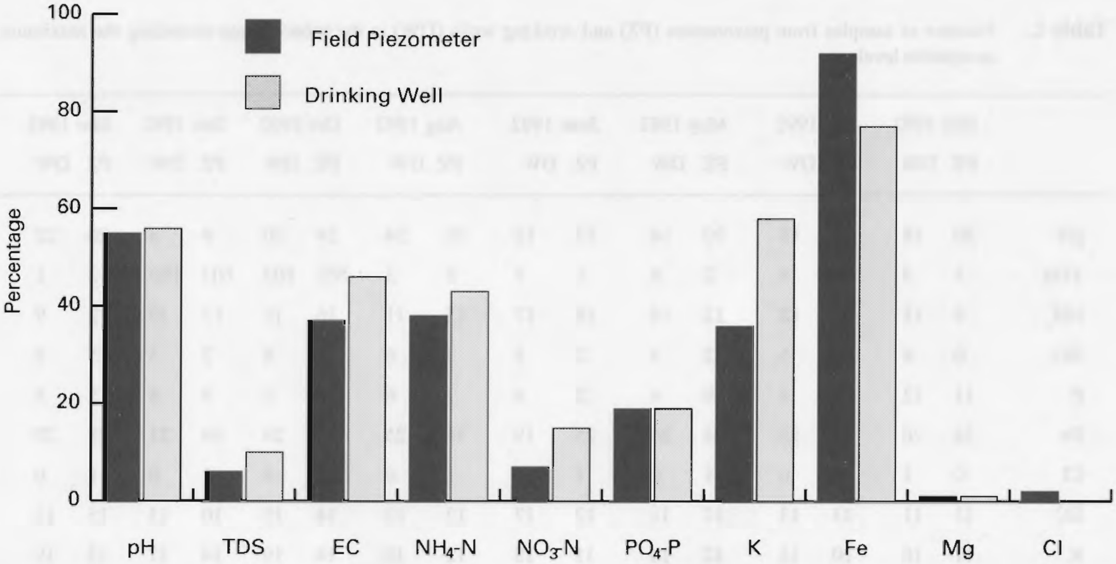

Figure 3. Percentage of water samples from piezometer and drinking well exceeding the Malaysian maximum drinking standards for inorganic parameters in tobacco agrosystem.

Results and Discussion

Chemical Survey of the Groundwater of the Kelantan Plain

It should be realised that high Fe and low pH are natural background characteristics in groundwaters in this region. It is unlikely that they have been influenced significantly by agriculture.

Tobacco agrosystem

Samples from piezometers and drinking wells that had various inorganic levels above the permissible standards are presented in Table 1. The percentages of water samples exceeding the Malaysian standards are shown in Figure 3. The major chemicals of concern are: NO_3-N, Fe, P, NH_4-N and K.

Nitrate contamination was found to be prominent and persistent in drinking wells. On average, 15% of drinking well samples were found to have NO_3-N concentrations above the maximum acceptable level of 10 mg/L. However, only 7% of the piezometer samples had such a high NO_3-N level. The highest level of NO_3-N detected in drinking wells was 30 mg/L. Samples from drinking wells at T7, T10, T14, T19, T21, and T28 exceeded the WHO standard of 10 mg/L for NO_3-N at one or more samplings. Figure 4 shows the trend in NO_3-N concentration with sampling times at five sample sites. The percentage of well samples which exceeded the drinking standard were: 77% for Fe, 43% for NH_4-N, 19% for P, 58% for K and 56% for pH. Most samples exceeded the limit of Fe, the percentage being somewhat larger for piezometers than wells. Most other pollutants had percentages somewhat higher for wells than piezometers (Fig. 3). The pH values were lower than acceptable levels in more than 50% of cases.

Rice agrosystem

In the rice agrosystem, Fe, P and NH_4-N were found to be the main pollutants. More than 90% of samples from piezometers and drinking wells exceeded the maximum acceptable Fe level of 0.3 mg/L (Fig. 5). In the case of NH_4-N, 31% of piezometers and 27% of drinking wells exceeded the acceptable level. However, a more widespread NH_4-N contamination in both piezometers and drinking wells occurred in July 1992 (Table 2). On average 4% of piezometers and 9% of drinking wells had P above the acceptable level of 0.2 mg/L. Almost all piezometers and drinking wells had a pH below the acceptable value of 6.5.

Table 1. Number of samples from piezometers (PZ) and drinking wells (DW) in the tobacco area exceeding the maximum acceptable levels.

	Mar 1992		Apr 1992		May 1992		June 1992		Aug 1992		Oct 1992		Dec 1992		Mar 1993	
	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW
pH	20	18	15	15	20	14	17	13	26	24	24	20	9	8	20	22
TDS	3	3	2	4	2	4	1	3	2	3	ND	ND	ND	ND	2	1
NH_4	9	11	7	12	12	14	18	17	13	11	16	11	17	10	12	9
NO_3	0	4	2	5	2	3	2	5	3	6	3	6	2	3	5	4
P	11	12	5	4	9	4	2	4	7	6	4	5	9	5	5	5
Fe	34	26	34	23	34	26	25	19	35	25	33	24	34	21	25	20
Cl	0	1	0	0	1	0	1	0	1	0	1	0	1	0	1	0
EC	13	11	13	13	12	11	12	17	12	17	14	15	10	13	15	13
K	11	16	10	14	12	18	11	18	12	18	14	19	14	17	15	19
Mg	0	0	3	1	0	1	0	0	0	0	0	0	0	0	0	0

Total number of samples: PZ = 35 and DW = 30, ND = not determined

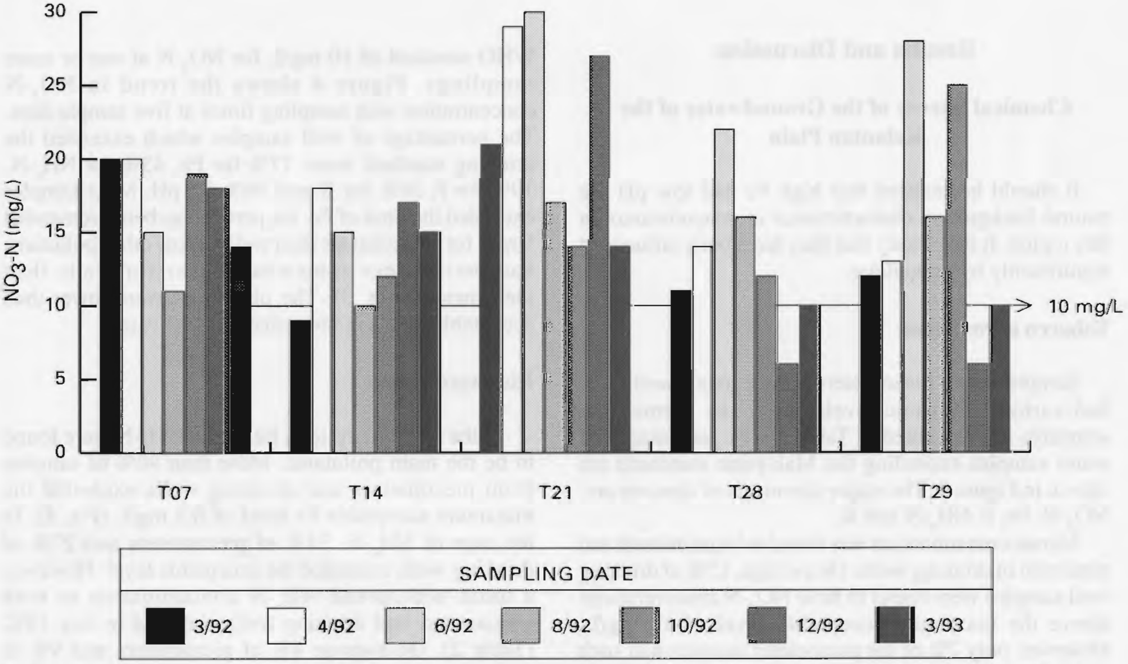

Figure 4. Measured NO_3-N concentration in drinking well at five locations in tobacco agrosystem where nitrate levels exceed the WHO drinking water standard.

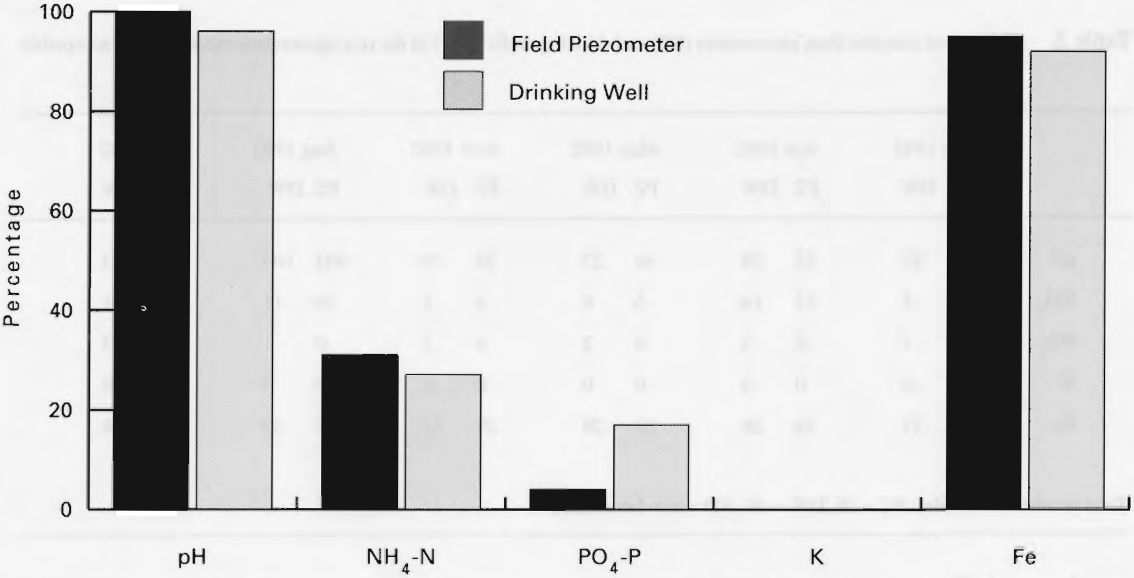

Figure 5. Percentage of water samples from piezometer and drinking well exceeding the Malaysian maximum drinking standards for inorganic parameters in the rice agrosystem.

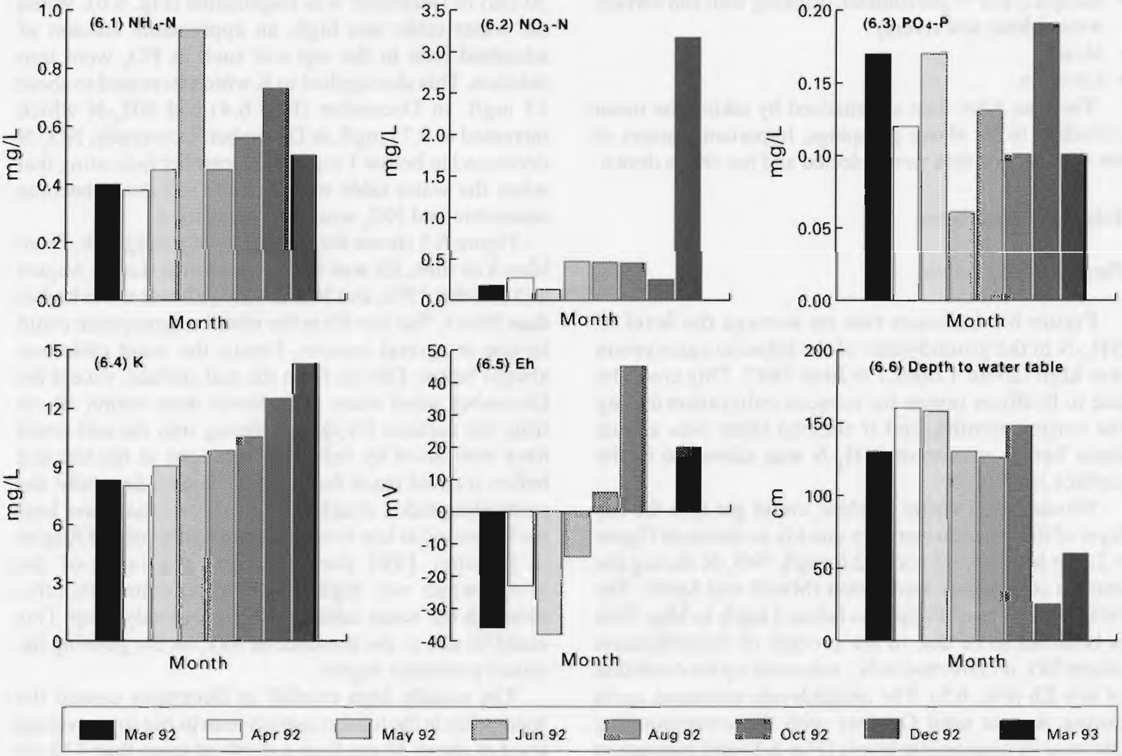

Figure 6. Variations of NH_4, NO_3, PO_4, K, Eh and depth to water table from the piezometers measurement of the groundwater in the tobacco agrosystem.

Table 2. Number of samples from piezometers (PZ) and drinking wells (DW) in the rice agrosystem exceeding the acceptable level.

	Mar 1992		Apr 1992		May 1992		June 1992		Aug 1992		Oct 1992	
	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW	PZ	DW
pH	35	29	35	29	34	27	34	29	ND	ND	ND	ND
NH_4	6	8	23	14	5	6	5	5	16	11	6	1
PO_4	4	4	0	3	0	2	4	3	0	1	0	3
K	0	0	0	0	0	0	0	0	0	0	0	0
Fe	35	27	34	28	33	28	29	25	35	30	31	26

Total number of samples: PZ = 35, DW = 30, ND = not determined

Inorganic Contaminants in Groundwater

The collected data were classified into various groups

- Agrosystem — tobacco and rice
- Sample types — piezometer, drinking well and surface water (drain and rivers)
- Month
- Location

The data were then summarised by taking the mean according to the above groupings. Important features of the summarised data were selected and bar charts drawn.

Tobacco agrosystem

Piezometer by month

Figure 6.1 indicates that on average the level of NH_4-N in the groundwater of the tobacco agrosystem was high (about 1 mg/L) in June 1992. This could be due to fertiliser inputs for tobacco cultivation during the earlier months, and it seemed there was a time lapse before maximum NH_4-N was adsorbed in the surface soils.

Nitrate being easily leached, could get into the top layer of the groundwater very quickly as shown in Figure 6.2; the levels were about 2.6 mg/L NO_3-N during the months of fertiliser application (March and April). The NO_3-N levels then dropped to below 1 mg/L in May. This is believed to be due to the process of denitrification where NO_3 is converted to N_2, enhanced by the condition of low Eh (Fig. 6.5). The nitrate levels increased again during August until October with the corresponding decrease in ammonium levels (Fig. 6.1) and increase in Eh. Nitrification is thought to have taken place during these months.

P concentrations were erratic with no interpretable trends (Fig. 6.3). However, in December 1992, the average P levels reached nearly 0.2 mg/L, the highest for the year. It is thought that the high water table (about 30 cm) in December was responsible (Fig. 6.6). When the water table was high, an appreciable amount of adsorbed ions in the top soil such as PO_4 went into solution. This also applied to K which increased to about 13 mg/L in December (Fig. 6.4) and NH_4-N which increased to 0.75 mg/L in December. Conversely, NO_3-N decreased to below 1 mg/L in December indicating that when the water table was high the soil could become anaerobic and NO_3 would be denitrified.

Figure 6.5 shows the variations of average Eh. From March to June, Eh was below zero while during August and October 1992, and March 1993, Eh was never higher than 50 mV. The low Eh in the tobacco agrosystem could be due to several reasons. Firstly, the water table was always below 120 cm from the soil surface, except for December when water table levels were within 30 cm from the surface. Oxygen diffusing into the soil could have been used by reduction reactions at the top soil before it could reach further into the soil. Secondly, the controlling redox couple — Fe^{3+}-Fe^{2+} — may have kept the Eh poised at low levels. During the months of August to October 1992 the Eh of the top layer of the groundwater was higher than the previous months, although the water table was comparatively deep. This could be due to the presence of NO_3 on Eh, pushing the mixed potentials higher.

The usually high rainfall in December caused the water table in the tobacco agrosystem to rise to an average level of about 30 cm from a depth of more than 130 cm (Fig. 6.6). The rise in the water table had in turn caused several chemical changes in the top soil which contributed to the changes of chemical constituents of the top layer of the groundwater. These were discussed earlier.

Drinking well by month

Figure 7.1 shows the average levels of NH_4-N were similar to that of the piezometers, with the highest level in June 1992. Generally the levels of NH_4-N in the drinking wells were higher than the piezometers except in December. The effect of fertiliser input as discussed for Figure 6.1 on NH_4-N levels in the water samples of the drinking wells is also clearly shown in Figure 7.1 indicated by the highest level in June.

The NO_3-N levels in the samples from the drinking wells were more than twofold higher than those from the piezometers (Fig. 7.2) except for the March 1993 level which was the same as the piezometer. The same explanation could be given to the variations for the levels of NO_3-N with months as in the piezometers.

The average Eh of the water samples from the drinking well is shown in Figure 7.3. It is shown that Eh was positive from March 1992 to March 1993 and overall, the values were higher than those of the piezometers. Nitrate could have played a more important role in influencing the Eh in the drinking well, considering the higher levels of NO_3-N detected (Fig. 7.2). Another important redox couple acting in the drinking well system was the O_2–H_2O. The interplay of the two redox couples may have kept the system poised at the Eh as shown.

Figure 7.4 shows the average water table levels in the drinking wells in the tobacco agrosystem. The variation trend was similar to that of the piezometers (Fig. 6.6) except that they were about 6 cm deeper. Although the water table in the drinking well was deeper than in the piezometer, the Eh was higher. This is because the wells were open to the atmosphere and there was no flushing out of the water before sampling as in the piezometers.

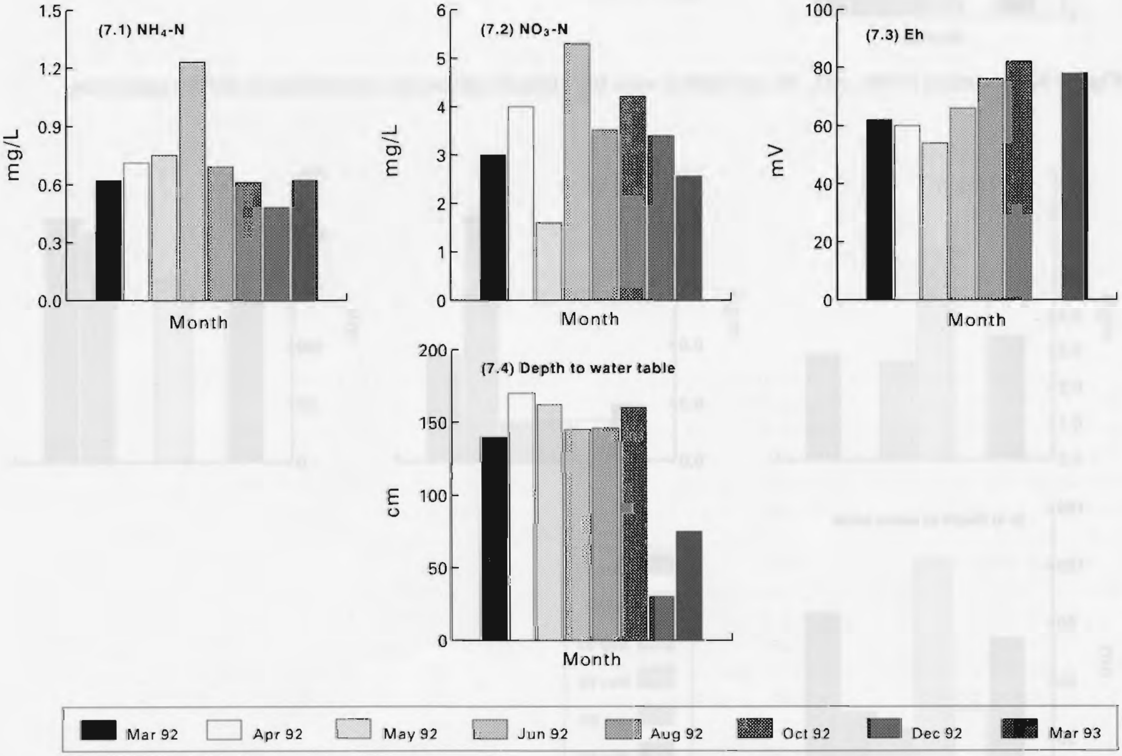

Figure 7. Variations of NH_4, NO_3, Eh, RP and depth to water table from the drinking well measurement of the groundwater in the tobacco agrosystem.

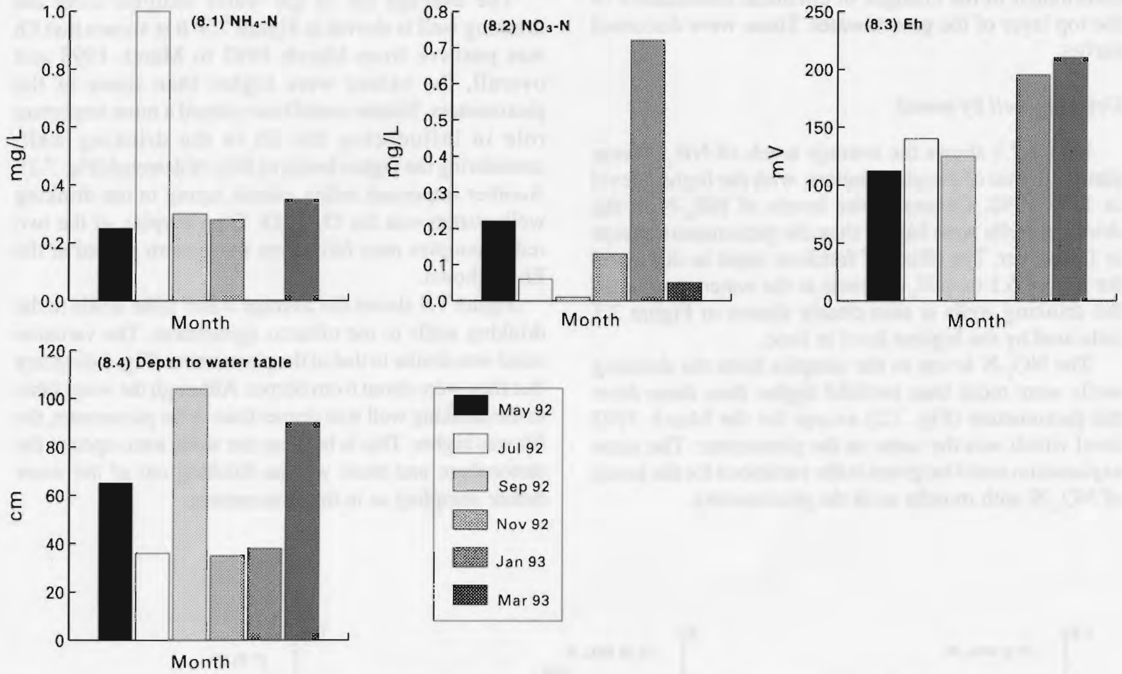

Figure 8. Variations of NH_4, NO_3, Eh and depth to water table from the piezometer measurement in the rice agrosystem.

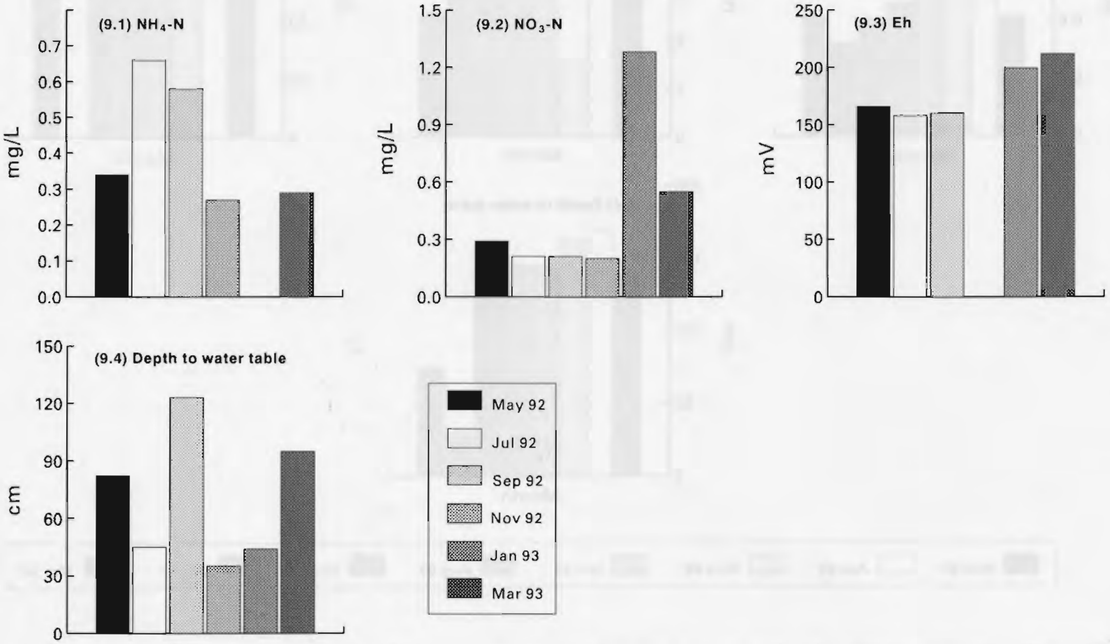

Figure 9. Variations of NH_4, NO_3, Eh and depth to water table from the drinking well measurement in the rice agrosystem.

Rice agrosystem

Piezometer by month

The average levels of NH_4-N were generally below the permissible limit of 0.5 mg/L, except in the month of July 1992 when the level was about 1 mg/L (Fig. 8.1).

Figure 8.2 shows the levels of NO_3-N by month. The levels were generally low, nevertheless, the levels in March and July 1992 were six times higher than the other months. This could be due to fertiliser inputs into the rice agrosystem. The lowest level was in January 1993 which could be due to dilution by high rainfall.

The Eh in the piezometers by months were above 100 mV (Fig. 8.3) and they had no bearing on the NH_4-N and NO_3-N levels. Water table levels were deep (>60 cm) in March and September 1992, and March 1993 (Fig. 8.4). The highest water levels were about 40 cm. Water table levels in the rice agrosystems are influenced by rainfall, irrigation and drainage. Water table would fluctuate with the irrigation and drainage schedules.

Drinking wells by month

The highest average NH_4 level in the water samples from drinking wells was observed in July 1992 (Fig. 9.1) which was slightly above the permissible limit. The occurrence of high NH_4-N levels coincided with that of the piezometers. In other months the levels were about 0.3 mg/L.

The nitrate levels were also highest in July 1992 (Fig. 9.2) although below the permissible limit, and lowest in March 1993 as in the piezometers.

The average Eh values of samples from the drinking wells were not much different from that of piezometers (Fig. 9.3).

The water table fluctuations in the drinking wells were similar to that of the piezometers (Fig. 9.4). Generally the levels were lower than those in the piezometers.

Detailed Analysis of Data from T-07 in the Tobacco Agrosystem

The data were initially classified into two data sets, (DS1 and DS2). DS1 includes those gathered from the grid piezometer, labelled A1 to D4. DS2 includes those gathered from piezometers and wells near the household toilet, labelled P1, P2 and W1. The data sets were analysed using the statistical package SAS.

Correlation analyses of DS1 and DS2

Correlation analysis was carried out on DS1 and DS2 to examine the interrelationships between the variables, and the correlation matrices are shown in Table 3. Electrical conductivity (EC) and total dissolved solids (TDS) of the groundwater in general (DS1) and of the groundwater around the household toilet (DS2) were very well correlated, with correlation coefficients of 0.94 and 0.93. EC and TDS were to a certain extent affected by depth of the watertable — they were inversely proportional with a correlation coefficient of -0.5. This shows that the water table movement has an appreciable influence on the chemical content (EC and TDS) of the groundwater contributed by NH_4, P and Fe. This was indicated by fairly high positive correlation coefficients which range from 0.5 and 0.6 for DS1 and from 0.5 to 0.9 for DS2. NO_3 and K has little contribution to EC and TDS in DS1, but NO_3 had quite a high negative correlation with TDS and EC in DS2 with correlation coefficients of -0.5 and -0.6. This could indicate that the toilet was a contributing source of NH_4 which was readily converted to NO_3-N when favourable conditions for oxidation of NH_4 prevailed. This process is indicated by a strong negative correlation between NO_3 and NH_4 in DS2 compared to a weak correlation in DS1. NO_3 also had a high positive correlation with Eh with a correlation coefficient of 0.7 in DS2 which indicates that NO_3 was readily reduced to N_2 under low Eh. High NO_3 levels in the groundwater around the household toilet were due to a higher rate of production rather than the reduced rate of dissipation of NO_3.

Changes of chemical variables affected by watertable variation with time

The water table in the tobacco agrosystem was greatly affected by the rainfall pattern and the above discussion reflects the importance of the effect of water table depth with respect to space and time during the study period. Figure 10.1 depicts the movement of the water table measured at the various sampling grid points and sampling time intervals during the study period. It is clear that the water table at each point in the area varied in a consistent pattern with time, although the magnitude of depth variations was not the same, as this depended on the surface morphology of the area. There were three incidences of flooding during the study period.

It has also been shown that Eh was linked directly with NO_3 and NH_4 and it is useful to examine the Eh variation with respect to space and time. Figure 10.2 shows the changes of Eh at each grid point with time. It is interesting to see a close relationship between the water table variation and the Eh changes with time. By comparing Figures 10.1 and 10.2 when the water table moved upward there was a simultaneous decrease of Eh which should be due to air being excluded from most of the pores in the capillary rise zone causing anaerobic conditions. The magnitude of the decrease of the Eh would depend on the amount of organic matter and the redox couple controlling the anaerobic reactions. During this condition NH_4 would remain unoxidised and NO_3

would be denitrified to N_2, Fe^{3+} would be reduced to Fe^{2+} thus releasing adsorbed P into solution. When the water table receded, water would drain out of most of the soil pores and be replaced by air which would make the soil aerobic again. Oxidation of NH_4 to NO_3 would commence. The NO_3 being mobile would move along with the rain or irrigation water into the groundwater pool. Fe^{2+} would be reoxidised to Fe^{3+} and ferric compounds, and P would be readsorbed. If the water table drops further, the concentration of oxygen in the soil air just above the groundwater would drop again, and the anaerobic reactions would start all over again, reducing the level of NO_3 and increasing the levels of Fe and P. The fluctuation of NO_3 levels in the groundwater at several grid points, as explained by the above mechanisms corresponded well with the depth of the water table and Eh variation (Fig. 10.3).

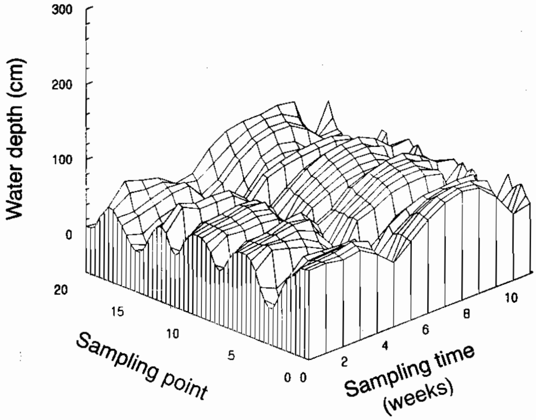

Figure 10.1. Variation of water depth with respect to space and time.

Chemical variables affected by watertable variations

From the above discussions it was shown that water depth from the ground surface had an influence on variables that were affected by Eh, such as NH_4, NO_3, P and Fe. The water table depth data showed

Table 3. Correlation matrix—DS1 data (piezometer).

	pH	Eh	EC	TDS	WT	NH_4	NO_3	P	K	Fe	Cl
pH		-0.27*** (9.99E-05)	ns	ns	-0.21** (0.003)	0.33*** (9.34E-07)	-0.21** (0.023)	ns (7.24E-12)	0.16* (0.007)	-0.45***	0.19**
Eh			ns	ns	ns	ns (2.50E 05)	0.29*** (0.046)	-0.14*	ns	ns	ns
EC				0.94** (7.04E-18)	-0.54*** (1.24E-16)	0.50*** (2.34E-14)	ns	0.55*** (-2.38E-17)	ns	0.52*** (1.82E-15)	0.79*** (-2.85E-17)
TDS					-0.50*** (1.70E-13)	0.59*** (7.56E-17)	ns	0.57*** (9.87E-17)	ns	0.51*** (3.22E-14)	0.87*** (3.49E-17)
WT						-0.43*** (8.43E-11)	ns	-0.51*** (5.41E-15)	0.19** (0.006)	ns	-0.49*** (4.32E-14)
NH_4							-0.21** (0.003)	0.34&*** (4.15E-07)	0.44*** (3.68E-11)	ns	0.64*** (4.37E-17)
NO_3								-0.22** (0.002)	ns	ns	ns
P									ns	0.49*** (4.09E-14)	0.47*** (1.24E-12)
K		ns = not significant value in parenthesis = P								ns	ns
Fe											0.30*** (8.30E-06)
Cl											

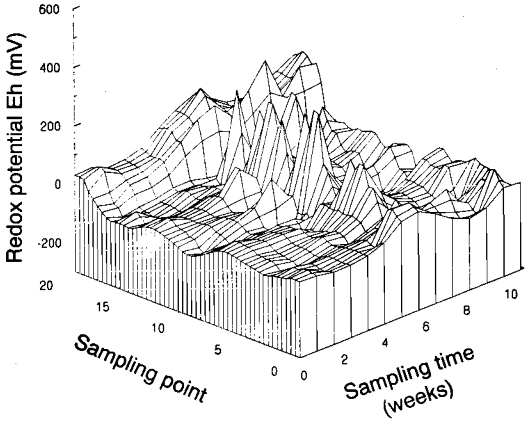

Figure 10.2. The changes of redox potential at each sampling point with time.

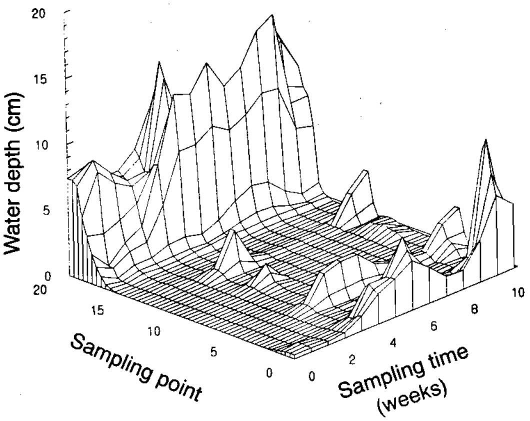

Figure 10.3. Fluctuation of NO_3-N levels in the groundwater at several grid points with time.

that they can be classified into those of high ground and low ground. Also the chemical data from samples collected around the household toilet had anomalous values compared to those collected at the piezometer grid points. Therefore, it is pertinent at this point to analyse the T-07 data according to high ground, low ground and around-toilet.

Figure 11 shows the average values of the changes of chemical variables in relation to water table variations with time, according to high ground, low ground and around-toilet classes. The trends of water table depth variations were similar for all the classes except at the low ground where there were incidences of flooding. The Eh trend for high ground and low ground were similar except the low ground redox potentials were more on the reduced side. The redox potential of the water samples around the toilet was always in the oxidised state after sampling time 3. NH_4 in the groundwater of the high ground had a downward trend, indicating nitrification with intermittent peaks when there was a negative change in the Eh. Conversely, there was a trend increasing for NO_3 in the groundwater in the high ground, although Eh had a strong influence on the concentration. NH_4-N concentrations were always high in the low ground (>1 mg/L). NO_3-N concentration in the low ground was always <0.22 mg/L probably due to denitrification. The NH_4 in the groundwater around the household toilet also had a decreasing trend but the NO_3 concentration was consistently higher at about 6 mg/L from the third sampling onwards. This does not support the explanation of the toilet contributing to the nitrate in the groundwater around the toilet. It seemed that groundwater in the high ground had a decreasing trend with time from 0.23 to 0 mg/L. This is contrary to that of the low ground which increased up to about 7 mg/L. The P levels around the household toilet increased up to about 0.25 mg/L. The P levels under the low ground followed more or less the level of Fe while there was a very poor relationship between P and Fe under high ground and around the household toilet.

Conclusions

- Wells currently used for drinking water in the tobacco and rice agrosystems in Kelantan exhibited widespread contamination with respect to inorganic pollutants. The problem was less threatening in the rice area.
- In the tobacco areas, 15% of drinking wells and 7% of field piezometers were found to contain NO_3-N above the acceptable level of 10 mg/L. Other pollutants of concern were Fe, pH and TDS. In the case of rice, Fe, NH_4 and P were of main concern.
- High Fe and low pH for these soils are existing conditions and have probably not been significantly affected by agriculture.
- In addition to fertiliser use, there was a likelihood of a significant contribution to nitrate and NH_4 levels from household septic tanks.
- Redox potential played an important role in

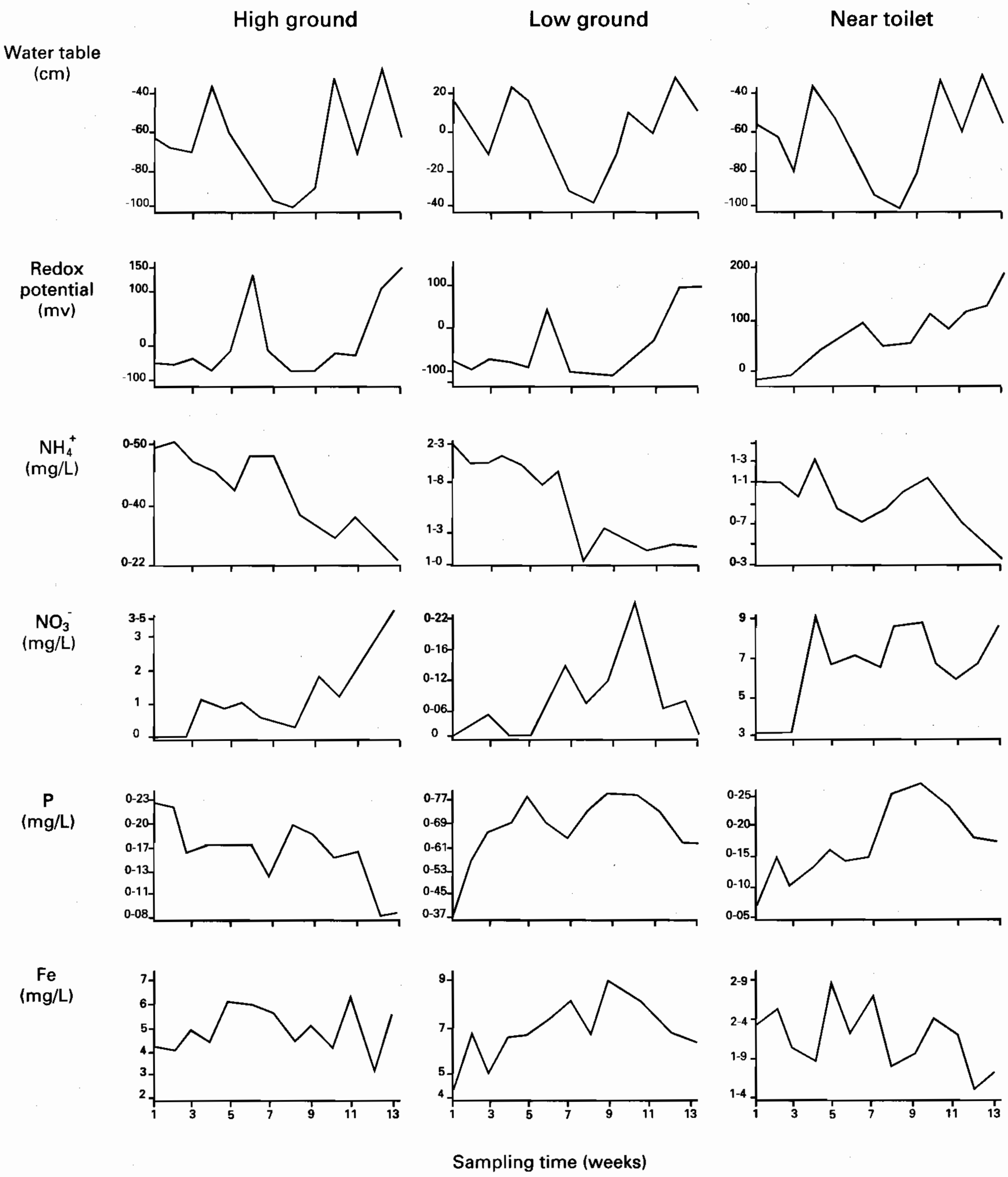

Figure 11. Changes of chemical variables in relation to water table variation with sampling time according to high-ground, low-ground and ground-toilet classes.

determining the levels of N and P.

- Water table depth had a substantial influence in determining the levels of N, P and K in the groundwater.

Acknowledgments

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Appendix 1. Drinking water standards.

	Malaysia	Australia	EEC	USA	WHO	Canada
pH	6.5–8.5	6.5–8.5	6.5–8.5	6.5–8.5	6.5–8.5	6.5–8.5
TDS mg/L	NA	1000	NA	500	1000	500
NH_4-N mg/L	0.5	NA	0.5	NA	NA	NA
NO_3-N mg/L	10	10	10	10	10	10
PO_4-P mg/L	0.2	NA	NA	NA	NA	NA
K mg/L	NA	NA	10	NA	NA	NA
Mg mg/L	NA	NA	30	NA	NA	NA
Fe mg/L	0.3	0.3	0.05	0.3	0.3	0.3
Cl mg/L	250	400	25	250	250	250
EC µS/cm	NA	NA	400	NA	NA	NA

NA: not available

Pesticide Residues in Water Resources of the Agricultural Plains of Kelantan

U.B. Cheah[1], M.L. Sharma[2], B.Y. Aminuddin[1], C.H. Mohammud[3] and M. Mohd Zain[4]

Abstract

A survey was conducted on pesticide residues in surface and groundwaters in rice and tobacco agrosystems of the agricultural plains of Kelantan in Malaysia. Groundwater is an important source of potable water among farmers in Kelantan. The results suggested there is evidence of pesticide residues as contaminants.

Samples taken from piezometers and wells generally showed similar results. The cyclodiene insecticide — endosulphan in the form of the beta-isomer, and oxidation product — endosulphan sulphate, was found as a ubiquitous contaminant. Residue levels ranged from <0.002 ng/mL to 4.4 ng/mL.

The organochlorine insecticides detected were lindane, aldrin, dieldrin, and DDT. Generally residues were low. The occurrence of the organochlorine residues was attributed to the continued although restricted use of the chlorinated pesticides.

Methamidophos and carbofuran residues were not detected although they are widely used in tobacco cultivation. 2,4-D was found in many samples in the rice agrosystem, but is not expected to cause a problem in view of its non-persistent nature. The preferential flow of paraquat in the soil profile may explain its presence in the groundwater at extremely low levels which did not have any environmental or toxicological significance.

A BROADSCALE survey was conducted to determine pesticide residues in surface and groundwaters in the rice and tobacco growing areas of the agricultural plain of Kelantan. The objective of the study was to obtain baseline data that could provide information for decision making on pesticide use, and establish directions for further study.

Rice is a staple crop in Malaysia. Use of pesticides, particularly insecticides and herbicides, is usually required for the successful cultivation of rice. Insecticides such as BPMC, carbofuran, endosulphan, and lindane are commonly used, while 2,4-D, paraquat and molinate are some of the common herbicides used for weed control. Many studies have shown that pesticide residue problems in rice grains do not arise from field applications of pesticides (Iman and Rejesus 1977; Cheah 1989; Cheah and Lum 1993). However, little is known about the environmental impact of these agrochemicals in the agrosystem.

Tobacco is an important cash crop in Kelantan. Pesticides are essential for the production of the crop. Methamidophos, mancozeb and carbofuran are commonly used.

Groundwater is an important source of potable water in Kelantan. It is estimated that 70% of potable water is derived from groundwater. Groundwater is the primary resource among farmers for rice and tobacco growing. Therefore, assessment of water quality for pesticide contamination is necessary. Although there are many possible sources of groundwater contamination, routine agricultural applications of fertilisers and pesticides are recognised increasingly as significant sources (Goodrich et al. 1991).

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Materials and Methods

Sampling Sites

Twenty-eight sampling sites in irrigated rice and tobacco growing areas were selected for the study. Greater numbers of sampling sites were located in areas where shallow groundwater was used for domestic consumption. The choice of sampling sites was based on location, site elevation, cropping system, source of irrigation and type of farm. The locations of the sites are shown by Ahmad et al. (these proceedings).

A piezometer was installed at each site, to collect water seeping from the surrounding paddy area. The characteristics of each site and details of the farming activities were recorded.

Sampling Procedure

Two litre water samples were taken from open household wells, field piezometers, open farm wells, regional primary and tertiary field drains, irrigation canals, and rivers. Sampling was conducted before, during, and after a growing season.

Water samples were collected in 2.5 L glass bottles and transported to the Pesticide Research Laboratories in Serdang. The samples were stored at 4°C and processed within 4 weeks.

Pesticides Examined

The pesticides examined were lindane, aldrin, dieldrin, endosulphan (alpha- and beta-isomers) and its metabolite — endosulphan sulphate, p,p'-DDT, o,p'-DDT and the corresponding metabolites DDE and DDD. The herbicides investigated were 2,4-D and paraquat.

Analytical Procedure

Organochlorine insecticide residues were analysed using the solvent partitioning method outlined by Lenore et al. (1989). Water samples were extracted using a methylene chloride/hexane mixture (15:85). A 500 mL sample was placed in a 1 L separating funnel and extracted with 200 mL of the solvent mixture. The aqueous phase was drained off and the residue extracted with another 200 mL of the solvent mixture. The solvent mixture was combined and dried over a column of anhydrous sodium sulphate. It was then concentrated to approximately 5 mL using a rotary evaporator. The hexane was evaporated to approximately 0.5 mL using a stream of N_2. Three mL of hexane was added, and again evaporated to approximately 0.5 mL. This procedure was repeated twice to remove final traces of the methylene chloride. Cleaning was not necessary as there was no interference in the gas chromatography process.

The extracts were analysed on a Perkin-Elmer Model 8500 Gas Chromatograph (Perkin-Elmer Corporation, Connecticut, USA) equipped with an electron capture detector for organochlorine insecticide residues. The capillary column used was 25 m long and packed with 3% phenyl methyl silicone gum, 0.32μm x 0.25μm **film** thickness. Injector and oven temperatures were maintained at 190°C and 200°C respectively. Oven temperature was programmed at a rate of 2°C per minute from 120°C to 250°C. Under these conditions, the minimum detection levels were 0.002 ng/mL for lindane, aldrin, dieldrin and endosulphan I (alpha-isomer) and 0.004 ng/mL for endosulphan II (beta-isomer), endosulphan sulphate, and DDT and its metabolites DDE and DDD. The results from the organochlorine analyses were confirmed using gas chromatography-mass spectrometry on a few samples chosen at random.

Methamidophos was extracted using solvent partitioning as described above. The extracts were examined using a gas chromatograph (Hewlett-Packard Model 5890) equipped with an N and P detector. The minimum detection limit was 0.01 ng/mL.

Carbofuran was determined by using solid phase extraction and liquid chromatography described in the method of Beauchamp et al. (1989). The extracts were examined on a high performance liquid chromatograph equipped with a UV detector. The detection limit was 2 ng/mL.

The analysis of 2,4-D and paraquat was carried out using a Millipore immunoassay kit. Standard solutions of 0.5, 10 and 100 ng/mL and a negative control were used. Three replicates were done for each sample. Optical density (OD) values for each well reaction on the microtitre plate were obtained using an Enzyme Linked Immuno Sorbent Assay (ELISA) plate reader at 450 nm wavelength. A calibration curve was constructed with the readings from the standard solutions, and was used to compute sample concentrations. The lowest detection limits for 2,4-D and paraquat were 0.5 ng/mL and 0.03 ng/mL respectively.

Results and Discussion

The study on rice and tobacco agrosystems in the Kelantan Plain revealed the presence of the cyclodiene insecticide — endosulphan, and organochlorine insecticides such as lindane, aldrin, dieldrin and DDT (Tables 1 and 2). Except for lindane, usage of organochlorine insecticides has been restricted, or banned in Malaysia. There was little difference in the types of pesticides found, and the number of samples detected and residue levels between groundwater from piezometer, and drinking well.

Table 1. Pesticide residues in water resources of rice agrosystems at three sampling times in the Kelantan Plain.

Time of sampling	Pesticides detected	No. of detected samples					% of detected samples exceeding EU standards	
		P[1]	W[1]	D[1]	IC[1]	R[1]	P[1]	W[1]
Jul 92 [2]	endosulphan	37	28	32	4	3	8.1	7.1
	lindane	21	16	16	1	2	38.1	56.3
	aldrin	19	18	18	2	1	26.3	33.3
	dieldrin	18	17	13	1	2	0	0
	DDT	6	9	4	0	0	0	0
	paraquat	11	7	8	1	1	0	28.6
	2, 4-D	10	5	11	2	3	80.0	100
Sept 92 [3] endosulphan	21	21	17	2	2	9.5	4.8	
	lindane	25	18	14	2	0	24.0	0
	aldrin	15	7	6	0	0	28.6	14.3
	dieldrin	26	21	22	2	0	3.8	0
	DDT	5	3	2	3	0	80.0	33.3
Feb 93 [4]	endosulphan	47	28	34	6	5	8.5	14.3
	lindane	3	3	3	1	1	0	0
	aldrin	15	10	4	1	1	0	10.0
	dieldrin	26	8	14	0	2	11.5	0
	DDT	9	3	8	3	3	11.1	33.3

[1] P = Piezometer W = Well D = Drain IC = Irrigation Canal R = River

[2] A total of 124 samples

[3] A total of 91 samples

[4] A total of 166 samples

Table 2. Pesticide residues in water resources of tobacco agrosystems at three sampling times in the Kelantan Plain.

Time of sampling	Pesticides detected	No. of detected samples				% of detected samples exceeding EU standards	
		P[1]	W[1]	D[1]	R[1]	P[1]	W[1]
Mar 92 [2]	endosulphan	29	26	-	-	20.7	19.2
	lindane	9	7	-	-	33.3	42.9
	aldrin	29	19	-	-	6.9	8.7
	dieldrin	NA	NA	NA	NA	-	-
	DDT	NA	NA	NA	NA	-	-
May 92 [3]	endosulphan	3	1	-	-	0	100
	lindane	20	16	-	-	80.0	87.5
	aldrin	29	23	-	-	6.9	8.7
	dieldrin	10	5	-	-	0	20.0
	DDT	12	6	-	-	16.7	33.3
Dec 93 [4]	endosulphan	0	1	2	0	0	0
	lindane	1	1	1	1	0	0
	aldrin	15	23	5	3	0	0
	dieldrin	4	4	3	0	0	0
	DDT	8	7	3	0	36.9	40.4

[1] P = Piezometer W = Well D = Drain R = River

[2] A total of 61 samples

[3] A total of 66 samples

[4] A total of 81 samples

Although organochlorine residues in groundwater have not been documented in Malaysia, residues of insecticides such as DDT, heptachlor and lindane have been reported in Malaysian rivers (Tan 1992). Chlorinated pesticides contaminating groundwater arising from normal agricultural use has also been reported in the USA (Ritter 1990) although the incidence is low.

The cyclodiene insecticide — endosulphan was found as a ubiquitous contaminant in the surface and groundwater of both rice and tobacco agrosystems. A high percentage of well and piezometer samples contained either alpha (endosulphan I) or beta-isomer (endosulphan II) of endosulphan, or its oxidation product — endosulphan sulphate. Residue levels ranged from <0.002 ng/mL to 4.4 ng/mL. It is interesting to note that although the alpha-isomer is more stable than the beta-isomer (Guerin and Kennedy 1992) the results show a higher number of samples contained the beta-isomer. Residue levels generally were also higher. The observation may be attributed to the loss of endosulphan I through volatilisation. Guerin and Kennedy (1992) found that through volatilisation, the alpha-isomer disappeared at a much faster rate than the beta-isomer, as a result of the former's higher volatility and lower water solubility. The ubiquity of endosulphan sulphate reinforces the conclusion that the oxidation product is very stable and considerably more persistent than the parent isomers (Miles and Moy 1979; Guerin and Kennedy 1992). The half-life of endosulphan sulphate in aqueous media is estimated between 140 and 184 days. Most of the samples analysed showed the presence of the endosulphan sulphate.

The occurrence of endosulphan in the rice growing areas was more widespread than in the tobacco areas. Higher residue levels were also noted in the rice agrosystem. This observation may be attributed to the continued use of endosulphan for controlling pests of rice. Usage of endosulphan in tobacco was common until officially withdrawn a few years ago (Yahya pers. comm.). A high adsorptive capacity of endosulphan in sandy loam was evident in laboratory studies utilising a radioisotope technique. More than 99% of the labelled insecticide was adsorbed onto sandy loam at equilibrium. Successive desorptions in the study showed values of 10.2%, 4.4%, 3.1% and 2.6% over four desorption processes (Cheah unpublished). The adsorption-desorption phenomenon may account for the occurrence of endosulphan residues.

Lindane residues were commonly detected in both agrosystems. The insecticide was more widespread in the rice agrosystem because of its continued use for the control of stem borers. However, residues levels were generally low (<0.002 ng/mL to 0.34 ng/mL) in spite of the persistent nature of the chemical. The persistency of lindane in soils has been reported in many studies conducted under field and laboratory condition (Fuhremann and Lichtenstein 1980; Cheah and Lum 1994). Its high volatility may result in the substantial loss of the insecticide from the locality of its application.

The occurrence of 2,4-D residues was not surprising considering its classification as a moderate leacher (Goodrich et al. 1991). However, the herbicide is not expected to be persistent as many studies have shown rapid degradation of 2,4-D in soils (Kaufman 1970; Helling et al. 1971). There was evidence of the occurrence of 2,4-D residues during the herbicide application only (Cheah, unpublished). The herbicide was detected in 30 out of 55 samples in the rice agrosystem. Residue levels ranged from 0.1 ng/mL to 44 ng/mL. None of the samples contained residues exceeding the Malaysian or Canadian standard of 100 ng/mL (Figs 1 and 2; Table 3). Thirty samples contained residues exceeding the EU standard of 0.1 ng/mL.

The occurrence of paraquat residues in the rice agrosystem was rather unexpected considering its property of strong adsorption to soil colloidal particles (Calderbank and Tomlinson 1969; Damanakis et al. 1970; Faust and Zarins 1969; Knight and Tomlison 1967) and that the soils in rice growing areas are mainly clayey. Residue levels detected were extremely low, ranging from 0.01 ng/mL to 0.13 ng/mL and should not pose any toxicological concern. The capability of detecting extremely low residues of paraquat in water (>0.03 ng/mL) is made possible through the progress achieved in the ELISA technique in recent years. The detection of paraquat residues in groundwater has not been reported until recently (Li and Wong 1992).

Paraquat residues in groundwater could be a result of preferential flow, which is the rapid transport of water and solute through a small portion of the soil volume which receives input over the entire inlet boundary (Guth and Mani 1991). The mechanisms that contribute to preferential flow include movement through structural voids, unstable flow of the entering fluid, or lateral dispersion of water into channels by partial surface clogging, or subsurface lateral flow.

Methamidophos, though widely used in tobacco cultivation, was not detected (<0.01 ng/mL) in any of the samples analysed. This was anticipated considering the non-persistent nature of the insecticide.

Carbofuran, used as a nematicide in the early stage of tobacco cultivation, was not detected (<2 ng/mL) in the samples collected in March 1992.

Aldrin residues were widespread in the tobacco agrosystem. Residues were found at a high level of 1.1 ng/mL during the sampling conducted in May 1992. Use of aldrin for termite control was very common in the tobacco growing areas until a few years ago. Aldrin has been reported to be extremely persistent with a half-life ranging from 5 to 10 years in soils (Edwards 1966). The occurrence of aldrin residues in the rice agrosystem was less widespread.

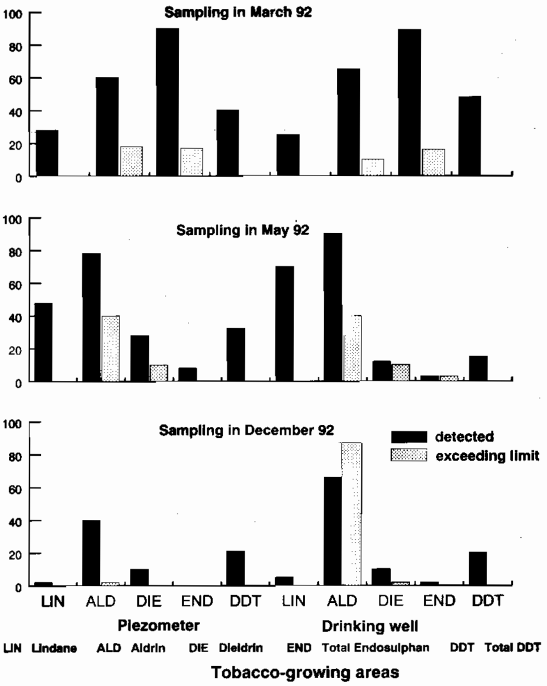

Figure 1. Percentage of detected pesticides and percentage of pesticides exceeding maximum permitted level (Malaysian Standard).

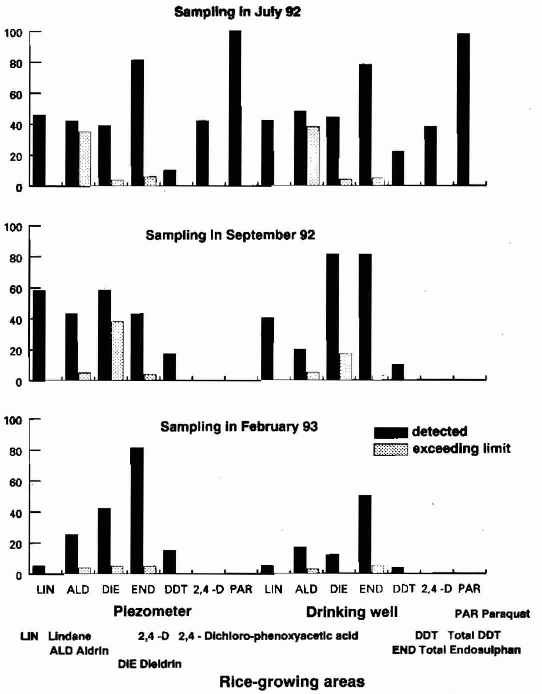

Figure 2. Percentage of detected pesticides and percentage of pesticides exceeding maximum permitted level (Malaysian Standard).

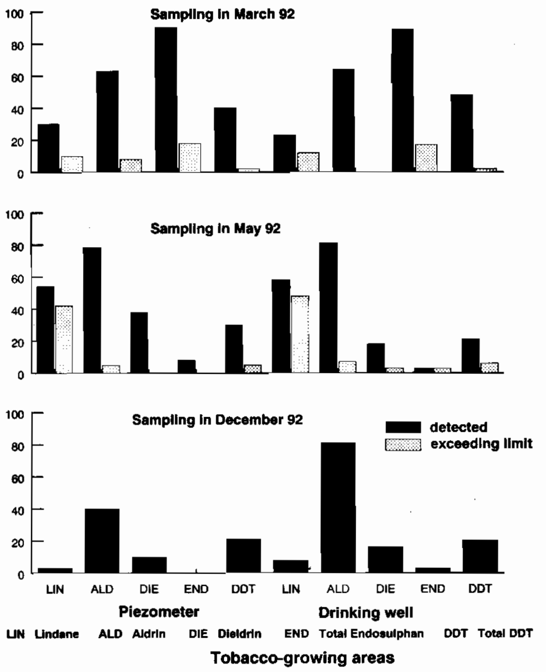

Figure 3. Percentage of detected pesticides and percentage of pesticides exceeding maximum permitted level (EEC Standard).

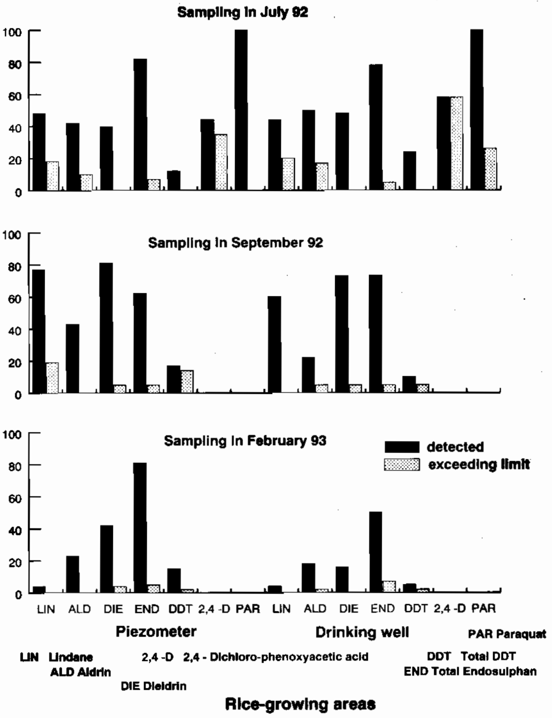

Figure 4. Percentage of detected pesticides and percentage of pesticides exceeding maximum permitted level (EEC Standard).

Table 3. Standards for drinking water.

	Malaysia	United States	Canada	Australia	WHO
Pesticide	Maximum acceptable limits ng/ml				
aldrin/dieldrin	0.03		0.7	1	0.03
carbofuran				30	
2,4-D	100	100	100	100	1
DDT	1		30	3	1
endosulphan				40	
lindane	3	0.4	4	100	NS

Source: Sayre 1988

The use of dieldrin in Malaysia is restricted to wood treatment. Residue levels found were generally lower than aldrin, and may have arisen from the past usage of aldrin, as dieldrin may be formed from aldrin in soils. This insecticide is reported as a moderate contaminator of groundwater.

The occurrence of DDT residues in groundwater was sporadic. Residue levels ranged from <0.004 ng/mL to 0.67 ng/mL for rice and tobacco agrosystems. The insecticide is not used as a pest control in the agrosystems, but is still commonly used for public health control purposes, especially for the control of vectors of malaria and dengue fever. Only a low percentage of well and piezometer samples contained DDT residues exceeding the EU standard of 0.1 ng/mL (Figs 3 and 4).

Conclusion

The presence of pesticide residues in groundwater of rice and tobacco agrosystems was evident. Endosulphan, including its oxidation product endosulphan sulphate, was found as ubiquitous contaminants. Cyclodiene was more frequently encountered in the rice agrosystem and at relatively higher residue levels. Among the organochlorines, lindane was commonly found in both agrosystems, while the occurrence of aldrin was more prevalent in the tobacco-growing areas. Sporadic detection of DDT in groundwater was observed. 2,4-D residues were commonly detected in the rice agrosystem in view of its appreciable mobility in the soil profile. The presence of low levels of paraquat residues may be attributable to preferential flow.

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Microbial Contamination of Water Resources in the Kelantan Plain

K.Y. Lum

Abstract

A monitoring program was undertaken to establish the extent of microbial contamination of water resources of the Kelantan Plain, Malaysia, as part of a comprehensive study on the impact of agricultural practices on water resources. This paper describes the broadscale survey and detailed field studies, and gives details of the extent and patterns of contamination in rice and tobacco agrosystems. The results showed that the groundwater under both rice and tobacco agrosystems was contaminated by coliform bacteria. Coliform numbers varied greatly in time and space, but there were large increases in colony forming units following the monsoon season. Data collected so far does not permit any conclusions to be drawn as to the exact source of this contamination.

THE KELANTAN PLAIN, an agriculturally significant area of about 0.2 million ha situated at the northeastern corner of Peninsular Malaysia, supports a population of approximately half a million people.

Two cropping systems — rice and tobacco, constitute the mainstay of agricultural activities in the area. The two agrosystems occupy about 33 000 and 13 000 ha respectively. Both crops are grown under irrigation. Paddy rice is grown on heavy riverine alluvial soils, and tobacco on the light-textured sandy coastal 'bris' soils.

In both agrosystems, the majority of farm households rely on water for domestic use from shallow wells in the vicinity of their farms. The relatively low water table (usually less than 2 m) together with the close proximity of households to farms, raises the likelihood of contamination of these wells as a result of agricultural activities. It is believed that the siting of toilets close to drinking water wells further aggravates the problem.

Project activities for the collaborative study of the impact of agricultural activities on the water resources of the Kelantan Plain included a monitoring program to establish the extent of microbial contamination of these water resources. The objective of this paper is to report the findings.

Material and Methods

Water samples for bacterial enumeration were sub-sampled from those collected at pre-determined intervals for organic analysis (Cheah et al. 1995; Ahmed et al. these proceedings). A 100 mL sub-sample was analysed for total coliform populations using the membrane filter method (APHA 1985). Pre-sterilised filters and the 'Endo' nutrient pad sets from Sartorius AG, Germany, were used throughout the study. In the enumeration of coliform populations, the following protocols were followed:

- nutrient pad sets were wetted with sterile distilled water in 50 mm disposable petri dishes,
- a 47 mm sterile membrane filter of 0.45μm pore size was placed in the filter holder,
- the sample was filtered, and
- the membrane was placed on the prepared nutrient pad set and incubated at 35°C.

Colony counts were taken after 24 hours of incubation. Where necessary, samples were pre-filtered through a 8μm filter before analysis.

Results and Discussion

Broadscale Surveys

Tobacco agrosystem

The broadscale survey for the tobacco agrosystem involved piezometer, well, irrigation and drainage sampling points at 30 sites (Ahmed et al. these proceedings). Sampling at these points was carried out from March 1992 until March 1993, after which sampling was reduced to 18 points at eight selected sites. Results obtained up until July 1994 showed that samples

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which did not register any coliform contamination at all ranged from 0 to 77%. The average percentage of samples which had 10 or less colony-forming units (CFU) per 100 mL, (recommended limit by World Health Organisation, WHO, 1983) was approximately 72% for the period indicated. Samples taken in March, May, and July 1993 registered the worst contamination figures when zero contamination percentages were 19, 0 and 0 respectively. Samples for July 1994 also registered a low zero contamination figure of 20% (Table 1).

Table 1. Distribution of coliform contamination levels for water samples from tobacco agrosystem.

Sampling date	Percentage contamination level					Number of samples
	0*	1-10	11-50	51-100	>100	
3/92	77	8	13	1	1	71
5/92	76	13	10	1	0	71
12/92	61	20	16	1.5	1.5	70
3/93	19	36	27	7	11	67
5/93	0	23	31	15	31	13
7/93	0	28	50	11	11	18
11/93	63	25	12	0	0	16
5/94	75	6	13	0	6	16
7/94	20	13	33	20	14	15

* Contamination levels expressed in CFU/100 mL

Rice agrosystem

The broadscale survey for rice included piezometer, well and irrigation and drainage points at 31 sites, beginning from July 1992. This was reduced to 16 points at eight sites after April 1993. Average percentage of samples which did not exceed 10 CFU per 100 mL was 73%. Samples taken from February until August 1993 registered low figures between 0 and 18% of zero contamination (Table 2).

Table 2. Distribution of coliform levels for water samples from the rice agrosystem.

Sampling date	Percentage contamination level					Number of samples
	0*	1-10	11-50	50-100	>100	
7/92	98	1	0	1	0	91
9/92	100	0	0	0	0	87
2/93	18	32	39	8	3	87
4/93	18	24	32	9	17	66
6/93	6	21	33	13	27	15
8/93	0	29	57	7	7	14
10/93	67	33	0	0	0	15
12/93	50	50	25	0	250	4
5/94	79	14	7	0	0	14
7/94	36	28	36	0	0	11

* Contamination levels expressed in CFU/100 mL

In general, results of the broadscale survey showed that wide fluctuations occur in total coliform populations in the groundwater of the tobacco and rice agrosystems. Therefore, comparisons between coliform counts in water samples from wells with those from field piezometers obtained over the experimental period do not allow any firm conclusions. For the tobacco agrosystem, it was evident that microbial pollution was serious (Barrow 1977; Cabelli 1977) during the period from March to July 1993. This trend was not evident in the 1992 results, perhaps due to infrequent sampling. The contamination figures for July 1994 tend to support this conclusion. With the rice agrosystem, higher levels of contamination were also observed for the period from June to August 1993. In both agrosystems, a rise in coliform levels was also evident following the monsoon season compared to the period preceding the onset of the monsoon.

Table 3. Average total coliform counts per sampling event for P-21 and T-07.

Rice Agrosystem		Tobacco Agrosystem	
Sampling date	Average CFU/100 mL	Sampling date	Average CFU/100 mL
16/11/93	0.9	13/01/93	2.8
02/01/94	36.5	18/01/93	6.8
19/01/94	29.1	25/01/93	14.2
31/01/94	17.8	01/02/93	15.3
16/02/94	11.1	09/02/93	16
02/03/94	30.9	17/02/93	31.0
23/03/94	40.7	24/02/93	23.6
30/03/94	16.4	03/03/93	27.8
12/05/94	2.3	09/03/93	38.1
18/05/94	16.4	15/3/93	0
24/05/94	34.9	22/03/93	41.1
31/05/94	11.1	30/03/93	32.8
07/06/94	9.4	27/04/93	24.1
14/06/94	18.3	27/06/93	3.4
28/06/94	2.3	24/10/93	0.6
05/07/94	34.5	03/01/94	8.6
13/07/94	28.3	24/01/94	0.8
23/07/94	7.8	08/02/94	0.4
27/07/94	0.6	23/02/94	3.4
03/08/94	11.3	09/03/94	1.1
09/08/94	12.6	21/03/94	4.9
15/08/94	16.9		
07/09/94	28.9		

Detailed Field Studies

Following the broadscale surveys, total coliform counts were also carried out on samples collected from points at the two representative sites selected for the tobacco (T-07) and rice agrosystem (P-21) selected for intensive studies (Sharma et al. these proceedings). Results of microbial enumeration were analysed to establish some understanding of the dynamics of microbial contamination at these two representative sites. Total coliform counts for individual sampling events averaged over 16 piezometer points for rice and tobacco agrosystems are given in Table 3.

The distribution pattern for average coliform counts for individual piezometer and well points over 23 sampling events from 1993 to 1994 are given for tobacco (Fig. 1) and for rice (Fig. 2). Overall, coliform contamination was high. In both agrosystems, water samples from the wells and adjacent piezometer sampling points registered high average total coliform counts (Table 4).

Table 4. Average total coliform counts for well and adjacent sites.

Rice agrosystem - R-21	Site	Well	P2	A3	B3	C3	B4
	CFU/100mL	23	12	18	23	30	30
Tobacco agrosystem - T-07	Site	Well	P1	C2	C3		
	CFU/100mL	24	26	12	21		

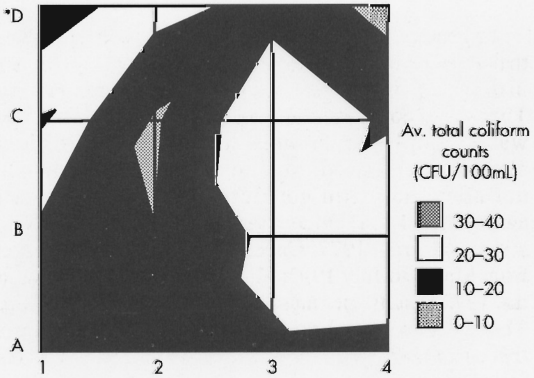

Figure 1. Spatial distribution of coliform population levels over T-07 sampling site.

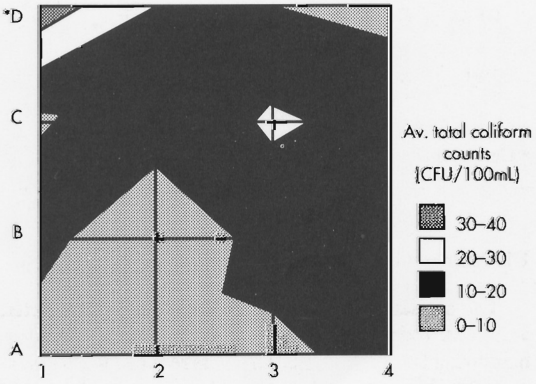

Figure 2. Spatial distribution of coliform population levels over R-21 sampling site.

* Axes refer to sampling points labelled as per overall site plan for T-07 and R-21 (see Sharma et al. these proceedings).

Conclusions

Results from the broadscale survey clearly showed that groundwater from the tobacco and rice agrosystems in the Kelantan Plain were contaminated. Fluctuations in total coliform levels varied widely over time and space, but there was an increase in levels immediately following the monsoon season. This was especially evident in 1993. Data from the intensive study in plots T-07 and R-21 confirm the contamination, as well as highlighting the tendency for high counts of CFU in drinking water wells.

While the present study establishes the quality of the groundwater in microbiological terms, no conclusions can be made regarding the source of the contamination. Coliform pollution can undoubtedly be caused by grazing farm animals and poultry. Intensive monitoring of coliform populations during changes in precipitation and groundwater levels including comprehensive information on water flow at selected sites could help provide answers with respect to point source contamination.

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Water and Nutrient Balance Study in the KADA Paddy Irrigation Scheme

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Abstract

A water and nutrient balance study was carried out over a rice growing period of 70 days to quantify components of water balance, especially deep percolation and seepage, with a view to measuring water-use efficiency and to assess the potential for groundwater contamination.

The average total water supply was 12.6 mm/d, of which only 36% was supplied by irrigation. Of the total water supplied, about 52% was transferred by surface drainage, 36% was lost through evapotranspiration and 12% through deep percolation and seepage. Average loss through seepage (0.8 mm/d) was comparable with deep percolation (0.7 mm/d) but it was lower than those reported in the literature. The extent of percolation and seepage is indicative of the potential to contaminate groundwater from surface applied agrochemicals.

Nutrient mass balances were used to quantify the relative sources and sinks of N, P and K. About 62% of the applied N during the season was lost via various mechanisms. The losses through leaching, surface run-off and ammonia volatilisation were estimated around 1.9, 4.6 and 7.3 kg/ha N, respectively. Loss through denitrification (~32 kg/ha N) was the most prominent. Most of the K loss was via surface run-off which was estimated around 9.6 kg/ha K or about 40% of the total applied.

IN THE KEMULU AGRICULTURAL DEVELOPMENT AUTHORITY (KADA) rice is cultivated twice a year. The main season normally falls between the wet months of October and March and is followed by an off season from April to September. Beside transplanting, direct sowing culture has been practiced in the KADA area since 1981. It has gained in popularity, and 75% of the area has now adopted the method. The water consumption is 10% higher than transplanting culture due to a longer growth period of about 20 days (Yashima 1984).

Paddy fields in the KADA irrigation scheme receive water via canal networks. Rotational irrigation is practiced within the Irrigation Service Area, and plot to plot irrigation within the Irrigation Service Unit normally comprises 20 ha of cultivated area. For a particular irrigation water used, increasing crop evapotranspiration (ET), results in decreasing stagnant field water. Water losses are due to seepage through and beneath ridges, percolation to the aquifer, and field surface drainage. Deep percolation may carry agrochemicals to the aquifer.

Field Water Requirement (FWR) for rice growth is a useful indicator for water-use efficiency. It is defined as the portion of water that has to be replaced by supplementary irrigation. It influences the frequency and amount of irrigation to be adopted under rotational irrigation. When derived empirically, it reflects the appropriateness of design criteria, or standards used, and the effectiveness of system operation and management.

The water-balance technique is used in the derivation of FWR and losses. All the parameters, such as irrigation, rainfall, standing water, evapotranspiration and surface drainage are obtained by direct measurement, while seepage and percolation are treated as residual parameters. In a plot to plot irrigation method, surface drainage is the water flowing to the next cultivated adjacent plot and then finally to a drain. Seepage and percolation constitute other water losses. In locations where boundary ridges are very strong and water levels

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on both sides of a boundary are maintained at a similar depth, seepage can be assumed to be minimal. However, some seepage underneath the ridges (bunds) is inevitable and can sometimes be substantial (Tabbal et al. 1992).

The objectives of this paper are to quantify components of water balance particularly percolation and seepage, and to quantify various components of N, P and K mass balance of a representative rice plot.

Study Area

A typical paddy farm was selected for the study. It is located in the KADA Lemal Irrigation Scheme at Kg. Pohon Tanjung, Mukim Alor Mas which is 10 km northwest of Pasir Mas Town (Fig. 1). The area has a comprehensive irrigation and drainage system network. The soils in the study area consist of Batu Hitam, Kg. Tepus and Sg. Amin series which are mainly of riverine origin. This soil is characterised by clay to loam texture with stratifications and is well to poorly drained.

The climate of the study area is strongly influenced by the monsoon wind with high rainfall occurring during the northeast monsoon, followed by a distinct dry period. The average rainfall and pan evaporation (Ep) over 11 years and the monthly rainfall during the study period are shown in Figure 2.

In the KADA rice growing areas nearly all the farmers are dependent on shallow wells for drinking water. Most of the wells are very close to toilets which may discharge into the shallow groundwater. A groundwater depth survey from 16 piezometers surrounding the study area (Fig. 3) shows there is a potential of horizontal flow to drinking wells due to the head differences. The trend can be seen from Figure 4 where the water table level of the drinking well was deeper compared to the surrounding piezometers during both dry and wet seasons. This situation may happen because of draw down of the well for household pumping. As a result, there is potential for polluting the household water supply.

Data Collection

Water balance

A water balance study was conducted in a small plot named C2. The layout is shown in Figure 3. To monitor water inputs, consumption and losses, 12 measuring devices were installed at the required places (Fig. 5).

The data collected were irrigation, rainfall, rice evapotranspiration, drainage and surface water level, evaporation and deep percolation. Irrigation input and

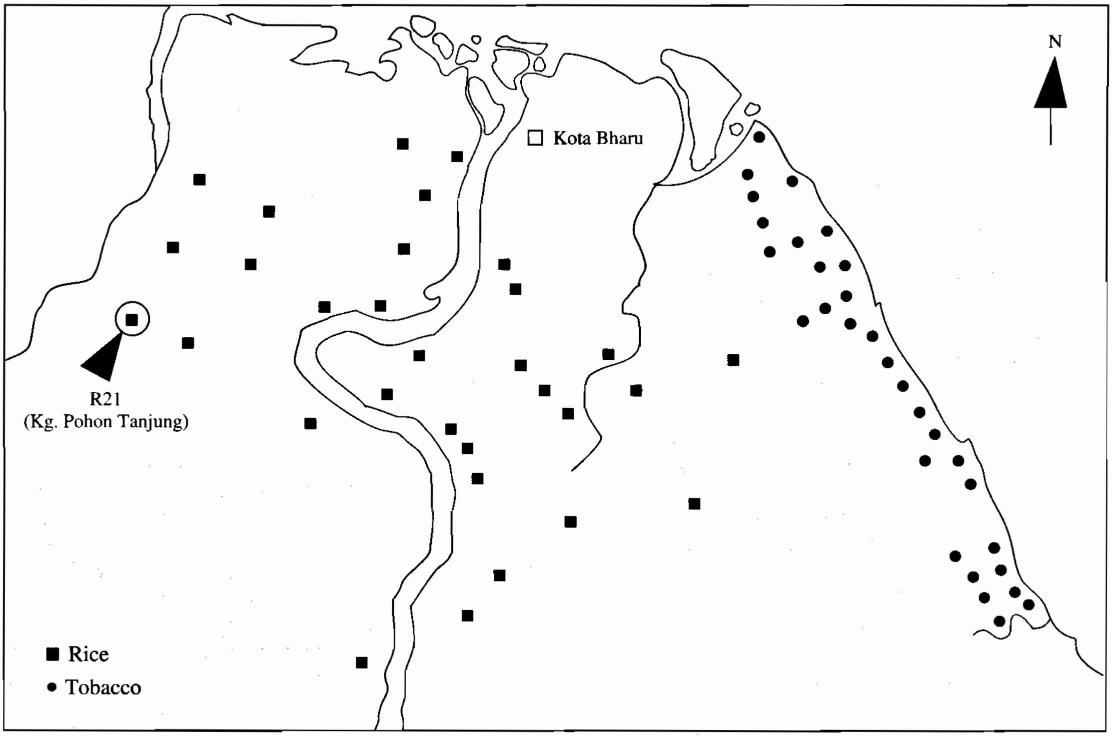


Figure 1. Location of water balance study (R21).

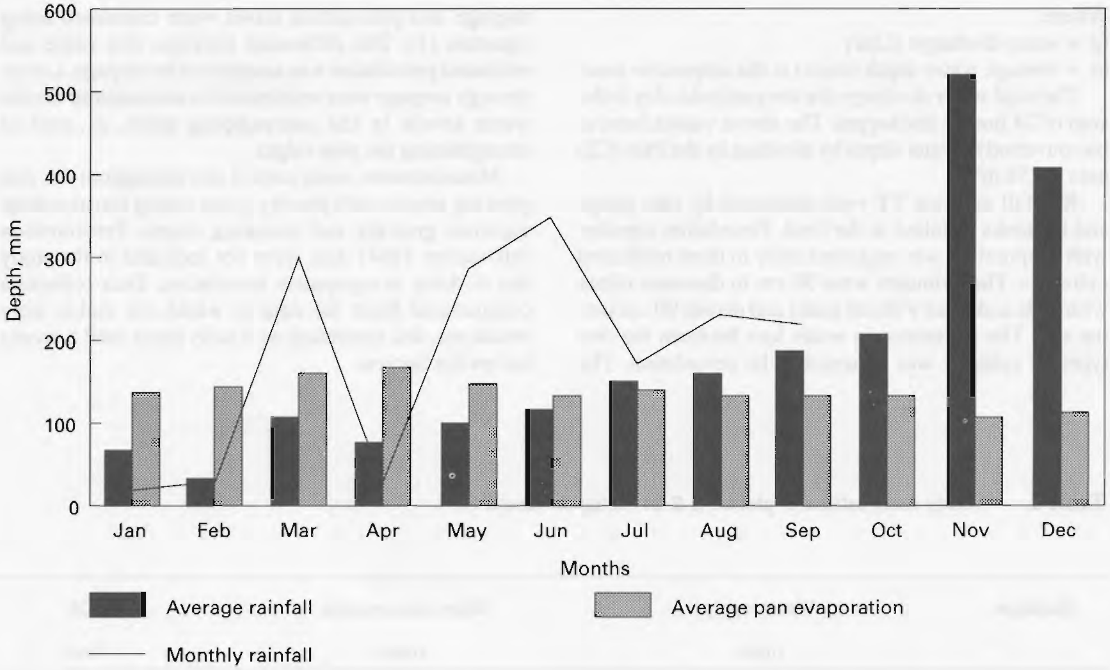


Figure 2. Average rainfall and pan evaporation at Kota Bharu over 11 years and the monthly rainfall during the study period.

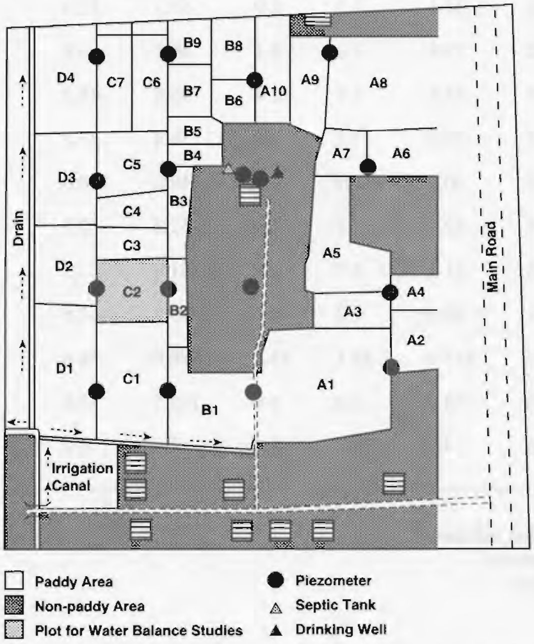


Figure 3. Plot layout for water balance study at R21.

surface drainage were measured using Parshall flumes at two locations each. Standing water depth, which was used to estimate a change in water storage, was measured by water level recorders placed at three locations across the plot. Rainfall, open pan evaporation and rice evapotranspiration were measured at one location each.

The water balance of the plot can be represented by:

$$DP + SP = IR + RF - SDR - ET - DS \qquad (1)$$

Where:

DP = Deep percolation
SP = Seepage
IR = Irrigation
RF = Rainfall
SDR = Surface drainage
ET = Evapotranspiration
DS = Change in field water storage

The water flow through Parshall flumes was regarded as the only source of irrigation supply into the study area. The amount of intake was calculated from automatic water level readings of Parshall flumes. Calculations of irrigation or surface drainage discharge (Q) through Parshall flumes were made by employing the following formula:

$$Q = 0.141\ ht^{1.55} \times 3600\ (L/hr) \qquad (2)$$

Where:
Q = water discharge (L/hr)
ht = average water depth (units) at the respective hour

The total water discharge for the particular day is the sum of 24 hourly discharges. The above values have to be converted to water depth by dividing by the Plot (C2) area (1758 m^2).

Rainfall and rice ET were measured by rain gauge and ET tanks installed in the field. Percolation together with evaporation was measured daily in three replicated cylinders. The cylinders were 30 cm in diameter (three with ends and three without ends) and driven 90 cm into the soil. The difference in water loss between the two types of cylinder was assumed to be percolation. The seepage and percolation losses were calculated using equation (1). The difference between this value and estimated percolation was assumed to be seepage. Losses through seepage were minimised by maintaining similar water levels in the surrounding plots, as well as strengthening the plot ridges.

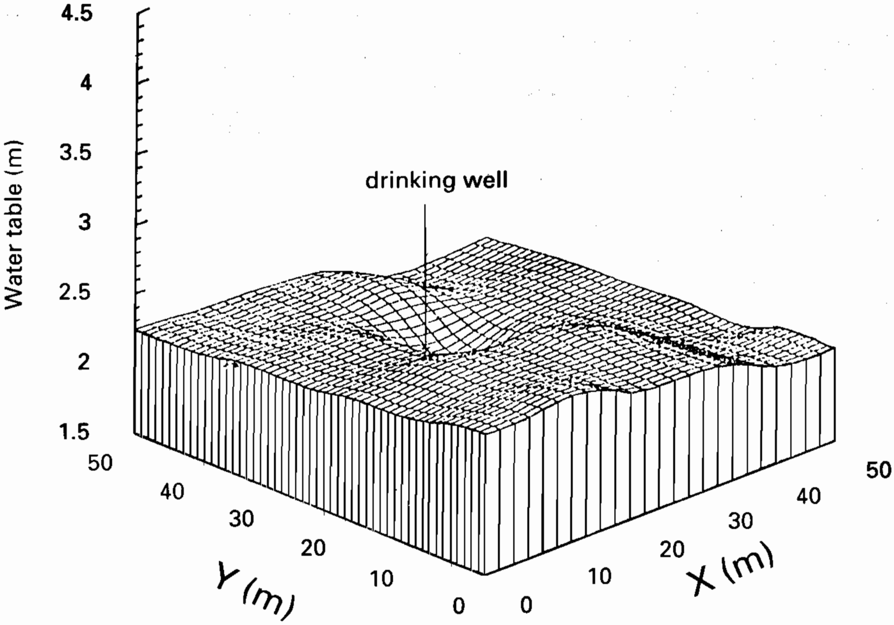
Measurements were carried out throughout the rice growing season with priority given during broadcasting, vigorous growing and maturing stages. Presaturation (off-season 1994) data were not included in the study due to delay in equipment installation. Data collection commenced from the date at which the fields were broadcast, and continued on a daily basis until 8 weeks before the harvest.

Table 1. Weekly water balance of plot C2 at R-21 during off-season 1994.

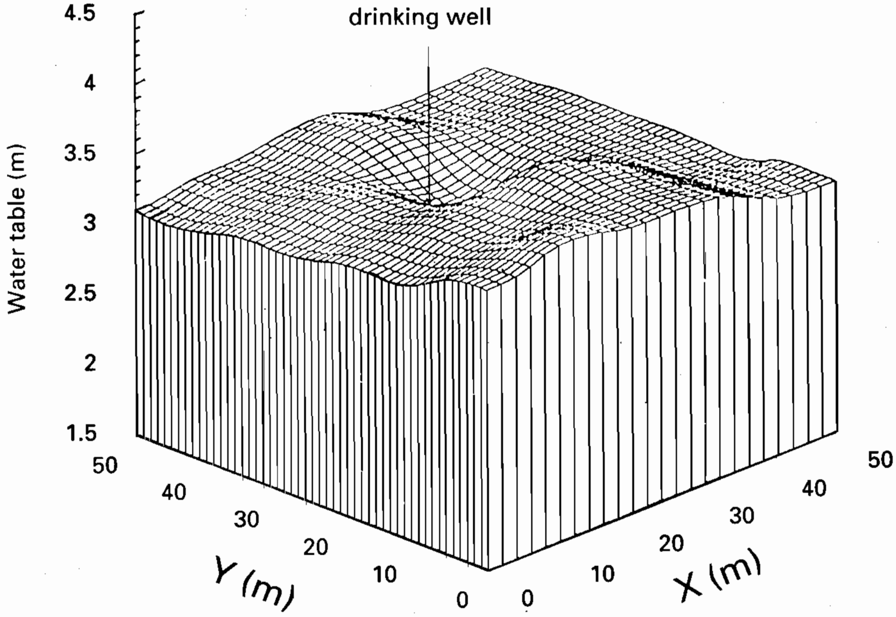
Duration	Water supply (mm)			Water consumption (mm)					DS (mm)
	RF	IR	Total	SDR	ET	DP	SP	Total	
18/6–24/6	66.2	45.3	111.5	118.7	31.7	10.0	-2.6	157.8	-46.3
25/6–01/7	82.0	155.9	237.9	102.0	27.5	12.3	3.8	145.6	92.3
02/7–08/7	8.0	1.4	9.4	23.2	34.1	1.5	2.9	61.7	-52.3
09/7–15/7	33.6	11.7	45.3	5.2	33.8	3.0	4.1	46.1	-0.8
16/7–22/7	51.2	1.4	52.6	40.9	41.6	1.9	12.4	96.8	-44.2
23/7–29/7	73.0	20.5	93.5	15.7	35.0	3.3	2.4	56.4	37.1
30/7–05/8	70.0	50.0	120.0	106.2	30.1	6.0	6.3	148.6	-28.6
06/8–12/8	20.4	1.4	21.8	1.4	22.5	1.2	7.0	32.1	-10.3
13/8–19/8	111.5	1.6	113.1	18.6	21.2	1.7	19.5	61.0	2.15
20/8–26/8	4.9	1.4	6.3	30.3	36.4	7.2	0.0	73.9	-67.6
Total	520.8	290.6	811.4	462.2	313.9	48.1	55.8	880.0	-68.6
%	64.2	35.8	100.0	57.0	38.7	5.9	6.9	108.5	-8.5
Av Daily	7.4	4.2	11.6	6.6	4.5	1.7	1.8	12.6	-1.0

RF	rainfall measured	ET	evapotranspiration measured
IR	irrigation supply measured	DP	percolation measured
SDR	surface drainage measured	SP	seepage computed
DS	change in field water storage measured		

Note: 1) Field water depth on 17th June 1994 was 65.3 mm and total water supply was 7.6 mm.
2) Field water depth on 26th August 1994 was 67.0 mm.



main-season



off-season

Figure 4. Water table level above mean sea level at R21.

Nutrient balance

A nutrient mass balance study was carried out for the plot C2. Water samples were collected at weekly intervals until 1 week after the last fertiliser was applied. Quality was measured in terms of N, P and K for irrigation water, rain, surface drainage and groundwater. Approximate mass balances were computed to define the relative sources and sinks of these nutrients.

Mass balance of N, P and K was attempted using measured and/or estimated values of the respective input/output components. The mass balance of N, P and K can be expressed as:

$$N_a + N_i + N_r = N_p + N_l \pm \Delta N_s \tag{3}$$

$$P_a + P_i + P_r = P_p + P_l \pm \Delta P_s \tag{4}$$

$$K_a + K_i + K_r = K_p + K_l \pm \Delta K_s \tag{5}$$

Where:

$N_a/P_a/K_a$ = Amount of N/P/K applied in fertiliser

$N_i/P_i/K_i$ = N/P/K in irrigation water

$N_r/P_r/K_r$ = N/P/K in rain water

$N_p/P_p/K_p$ = N/P/K uptake by rice plant

$N_l/P_l/K_l$ = N/P/K loss

$\Delta N_s/P_s/K_s$ = change of N/P/K in soil storage

N Balance: For N balance, the main components of loss are leaching, denitrification, ammonia volatilisation and surface run-off. All these components were measured or estimated, except the loss through denitrification, and changes in soil storage.

P/K Balance: The amounts of P/K in irrigation, rain and drainage were calculated using mean concentration of P/K in the water sample collected and the volume of water input/output for plot C2 (Table 2). The amount of P/K uptake by rice plants was estimated using the composition of P/K on annual production of plant materials. The measurements of changes in P/K soil storage were not attempted.

Table 2. Nitrogen concentrations in irrigation (IW), drainage (DW) and rain (RW) water at R-21.

Sampling No.	NH_4-N + NO_3-N (mg/L)		
	IW	DW	RW
S1	0.09	0.22	nd
S2	0.08	0.04	nd
S3	0.04	0.04	nd
S4	0.04	0.06	nd
S5	0.14	0.14	0.70
S6	0.08	0.13	0.34
S7	0.07	0.20	1.08
S8	0.32	0.68	0.64
S9	0.28	0.24	0.52
S10	0.24	0.18	0.42
S11	0.14	2.80	0.56
S12	0.12	0.36	0.54
S13	0.15	0.34	0.44
Mean	0.14	0.42	0.58

nd = not determined

Total volume:

Irrigation water	512.3 m^3
Rain water	998.2 m^3
Drainage	814.3 m^3

Results and Discussion

The Cropping Schedule

The progress of the cropping schedule during the observation period is shown in Figure 6. The area was directly seeded on the 3rd June, 1 month later than the recommended schedule. Irrigation water was supplied from the 1st April until the end of July, while the supplementary irrigation to the study plot started only

Table 3. Approximate partitioning of residual N (kg/ha N) into soil storage and various losses through leaching, ammonia volatilisation, surface run-off and denitrification.

Total residual N	Change in soil N storage	Leaching loss	Ammonia loss	Surface run-off loss	Denitrification loss
81	35	1.9	7.3	4.6	32.2

on the 16th June (i.e. 13 days after sowing). From the 3rd to 15th of June rainfall was the main contributor. The crop was harvested in the middle of October 1994.

Water Balance

Irrigation and surface drainage discharge

As the farmers practice 'plot to plot' irrigation, water supplied to plot C2 had to be diverted to down stream plots. The relationship of water supply and surface drainage in plot C2 is illustrated in Figure 7. A portion of water supply was released to the next plot as surface drainage, but at the same time field water depth had to be maintained at a suitable level. The difference between the water supply and surface drainage is the quantity of water remaining in the plot, and this contributes to ET, percolation, seepage and water storage.

The weekly components of water supply and water consumption are summarised in Table 1. The duration of the continuous irrigation during crop growth was only 70 days (18th June to 26th August 1994). The highest weekly irrigation supply occurred on the 4th week after sowing while the rest was dominated by rainfall. The total irrigation water supply at the study area during the period was 290.6 mm, while the total water supply (irrigation water plus rainfall) was 811.4 mm.

Standing water depth

The fluctuation of standing water depth (average of three observations) is shown in Figure 8. The average standing water depth during crop growth period was 70 mm. It took about 3 days to inundate the plot to 40 mm depth. The duration was long, due to uneven field levels especially in up stream and lower stream portions.

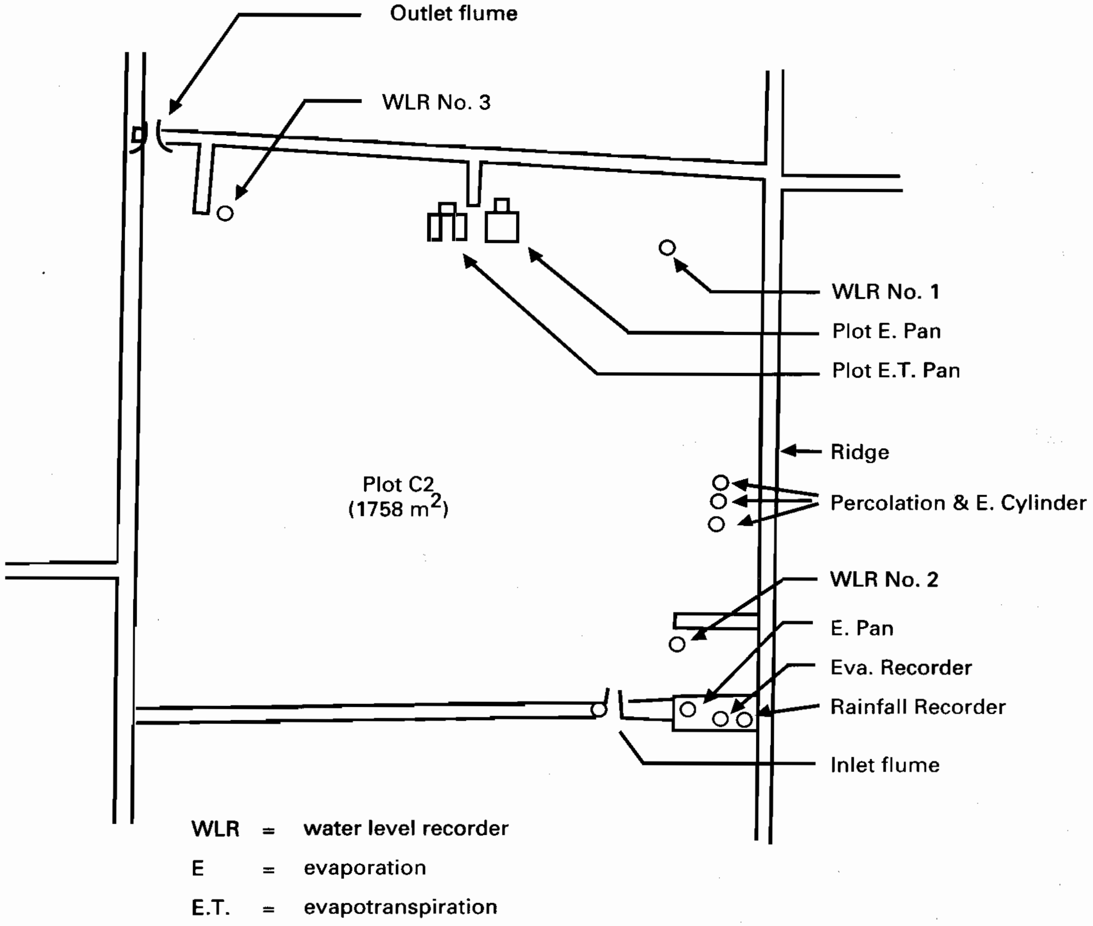
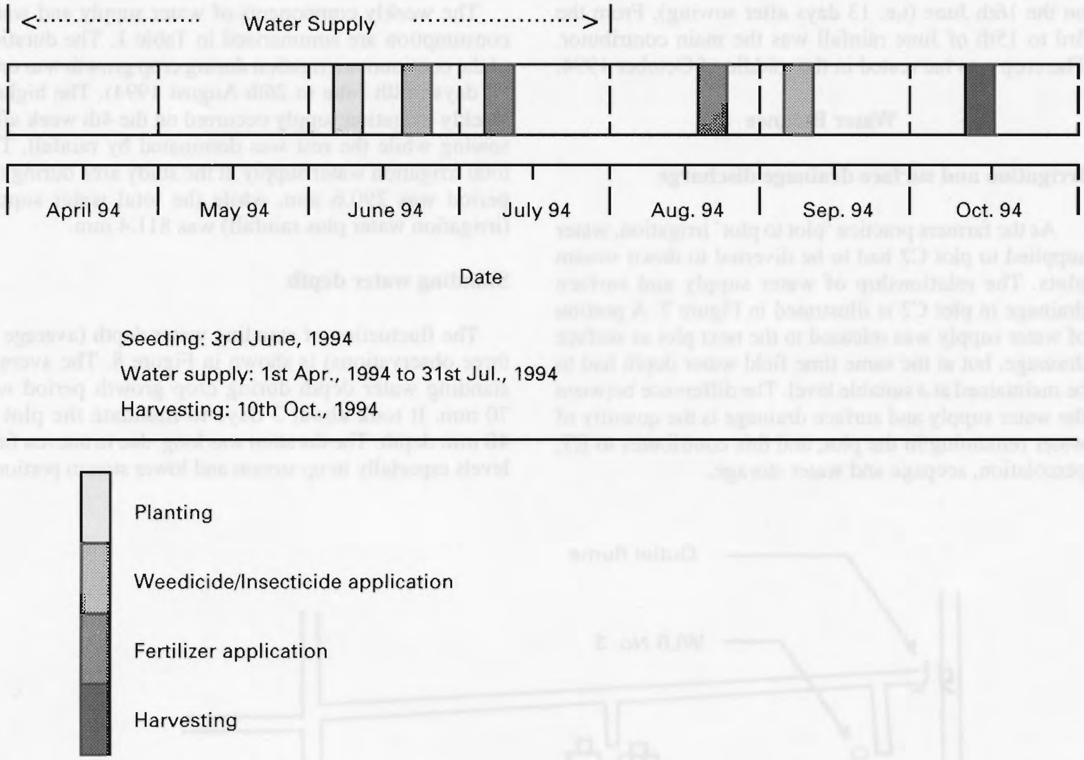


Figure 5. Plan showing location of the sampling points at R21.



Seeding: 3rd June, 1994
Water supply: 1st Apr., 1994 to 31st Jul., 1994
Harvesting: 10th Oct., 1994

Figure 6. Cropping Schedule at R21.

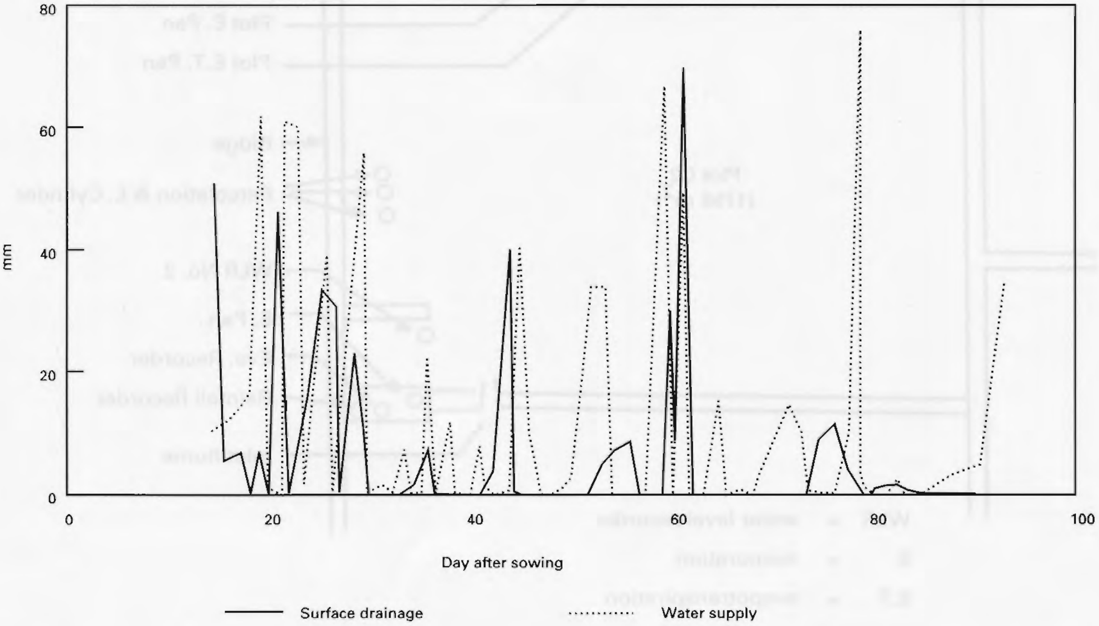


Figure 7. Daily water supply and surface drainage at Plot C2.

Water balance during crop growth

The water balance data (Table 1) shows that the total water input for 70 days (during 2 weeks after broadcasting to 7 weeks before harvesting) was 811.4 mm. Of this, 290.6 mm or 36% was supplied by irrigation and the other 520.8 mm or 64% by rainfall. Of this total water supplied, 314 mm or 36% was lost by evapotranspiration, 462 mm or 52% was transferred by surface drainage, and 104 mm or 12% was lost by deep percolation and seepage. About 69 mm or 8.5% was derived from stagnant water depth.

The designed demands for water supply were 883 mm as quoted in the KADA feasibility report. In comparison with the designed demands, the observed water supply was slightly less due to a shorter period of analysis, and would have been higher if this analysis was continued until 2 weeks before harvest. In the practice of direct seeding, the excessive water consumption normally occurred during presaturation and prior to broadcasting where standing water is drained out of the field, then refilled back after seed broadcast.

The average daily water supply during the rice growth period was 12.6 mm/d. It was estimated in the KADA feasibility report to be 9.2 mm/d. Actual water used at a similar area (MUDA) was about 10 mm/d (Yashima 1982). Therefore, 12.6 mm/d from the study plot is a reasonable value. In the season, the observed ET was 4.5 mm/d and percolation and seepage was about 3.0 mm/d, and in the main season 2.7 mm/d. It appears that seepage losses observed are much lower than those observed by others. In the MUDA area, Yashima (1984) observed seepage losses which ranged from 4.3–12.2 mm/d. In the Philippines, Woperies et al. (1994) reported very high seepage/percolation losses (4–36 mm/d) which were primarily due to seepage as percolation losses were <4 mm/d. These losses were much higher than evapotranspiration losses. In this study, the rate of percolation decreased from 2.3 mm/d at 3 weeks after broadcast to 2.0 mm/d or less (Fig. 9). The average percolation rate for 70 days of about 0.7 mm/d at the study plot is comparable to that of 0.9 mm/d estimated for MUDA area (Kitamura 1987).

Figure 10 shows the correspondence of weekly ET and Ep. The ET/Ep values of >1 for rice are not uncommon (e.g. Yashima 1984; Woperies et al. 1994). For direct seeding Yashima (1984) reported average ET/Ep values of 1.04 for first 50 days after sowing (DAS), 1.35 for 50 to 100 DAS, 1.46 for 100 to 120 DAS, and 1.08 for 120 DAS until harvest.

Nutrient Balance

Plot C2 received three applications of fertiliser. Applied fertiliser varied in N, P, K content. In the first application fertiliser had N: P_2O_5:K_2O ratio of 17:20:10, and in the second, it had a ratio of 63:20:10. Only N fertiliser at the rate of 46 kg/ha N was applied during the third fertiliser application.

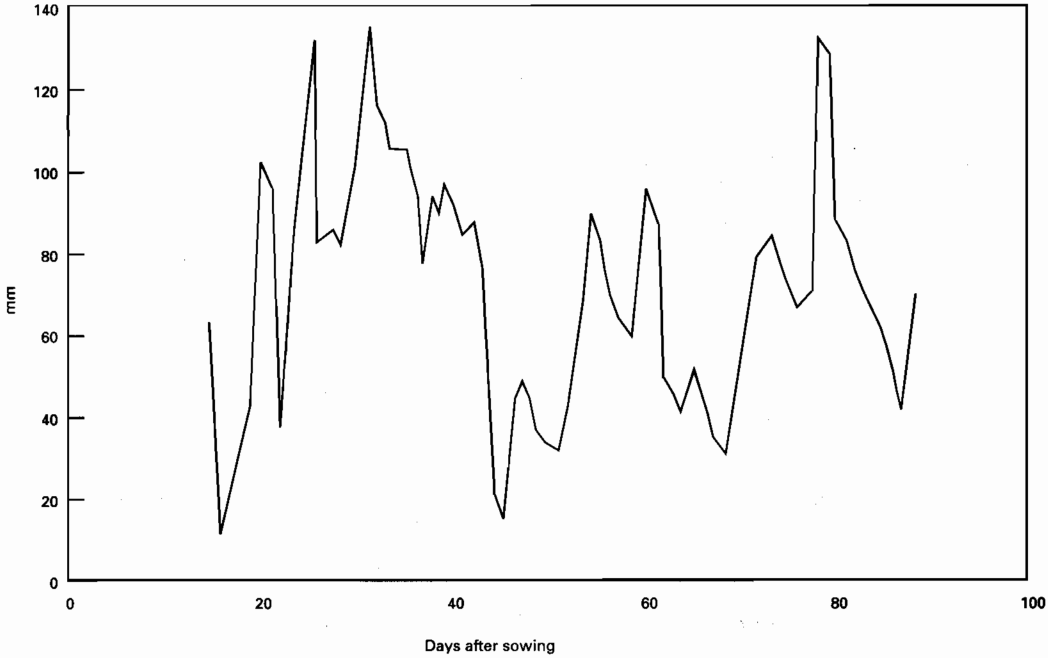


Figure 8. Daily fluctuation of surface water level in Plot C2.

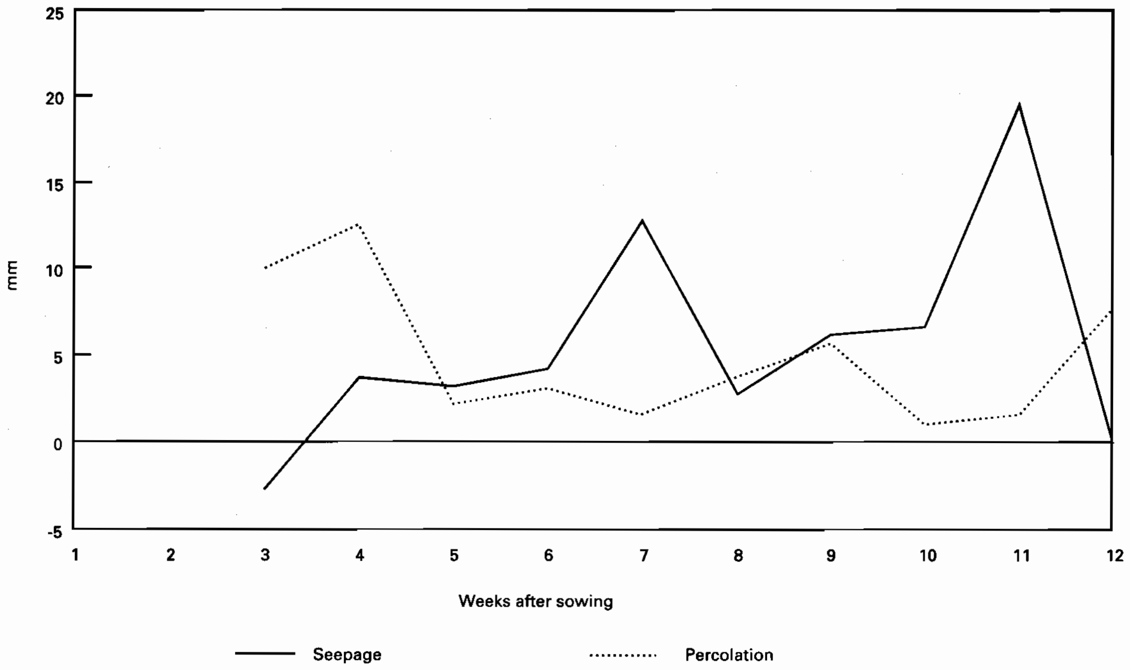


Figure 9. Weekly percolation and seepage at plot C2.

Figure 10. Weekly evapotranspiration and pan evapotranspiration at plot C2.

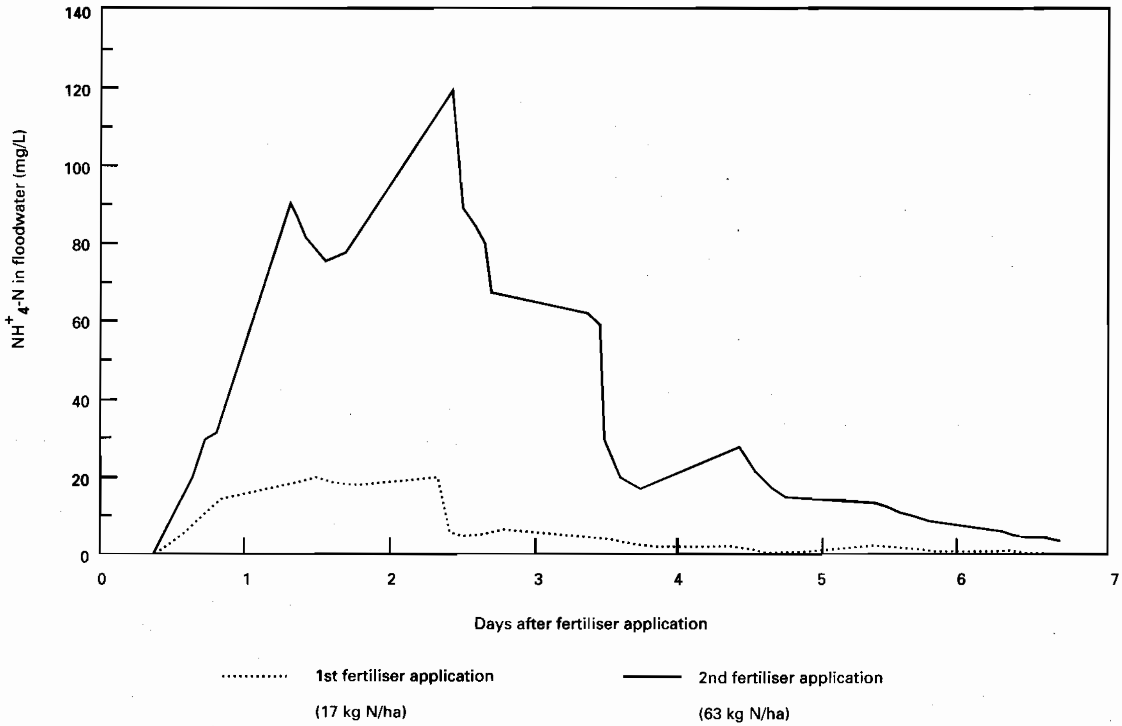


Figure 11. Concentration of NH^+_4-N in floodwater following application of N fertiliser.

Nitrogen

The composition of irrigation, drainage and rain waters with respect to N content, measured on several occasions is presented in Table 2. The amount of fertiliser N (Na) applied to plot C2 during the growing period was 126 kg/ha N. Rain water (Nr) supply was around 2.6 kg/ha N, while irrigation water (Ni) contributed only 0.3 kg/ha N (Table 3). Thus the total input of inorganic N was 124.8 kg/ha N. N uptake (Np) by rice plants was estimated around 48 kg/ha N (Aziz et al. 1987). Using these data the residual N was calculated to be 81 kg/ha N or 65% of applied N.

This residual N can either add to the soil N storage, or can be lost from the system through the processes of leaching, surface run-off, ammonia volatilisation and denitrification. Approximate estimates of these components for plot C2 are presented in Table 3. Based on measurements, the amount of N loss through surface run-off amounted to 4.6 kg/ha.

Concentration of NH_4-N in the floodwater is an important parameter for estimating loss through volatilisation. Change in NH_4-N concentration in floodwater after fertiliser application at plot C2 is shown in Figure 11. It shows that NH_4-N build up was rapid, reaching a maximum value on the second day. After 6–7 days the concentration decreased to very low values. It was observed that for the high N application rate (second fertiliser application) the concentration of NH_4-N in the floodwater was higher. However, the duration was only 2 days longer than the lower N rate (first fertiliser application). Calculated ammonia volatilisation loss was 0.85 kg/ha N (5%) for the first fertiliser application, 4.4 kg/ha N for the second application, and 2.0 kg/ha N (5.6%) for the third application. Thus the total loss of N through ammonia volatilisation was 7.3 kg/ha N.

Only 1.9 kg/ha N was lost through leaching. Apart from loss through ammonia and leaching, additional N loss occurs via denitrification. However, this was not measured in the present study. If the increase in N stored in soil was taken as 35 kg/ha N (Aziz et al. 1987) N loss through denitrification would amount to around 32 kg/ha N.

Phosphorus and potassium

Plot C2 received two applications of P fertiliser at the rate of 40 kg/ha P_2O_5; this is equivalent to 17.44 kg/ha P. The amount of P supplied by rain and irrigation water was less than 0.1 kg/ha P. Most applied P is taken up by the rice plants, and is estimated at around 14.2 kg/ha P. Thus the residual P is calculated as 3.2 kg/ha P. Of this only 0.1 kg/ha P was lost through surface run-off. Leaching accounted for about 3 kg/ha P loss. The change in soil P storage was estimated to be small (0.1 kg/ha).

The total input of K into the plot was 23.4 kg/ha K. Rain and irrigation waters supplied about 5.2 kg/ha K and 1.6 kg/ha K respectively. The rest came from fertiliser application which was at the rate of 20 kg/ha K_2O, or equivalent to 16.6 kg/ha K. Rice plants take up about 73.5% of applied K or 12.2 kg/ha K. The total residual K was thus around 11.2 kg/ha K. Of this 86% or 9.6 kg/ha K was lost through surface run-off via leaching. The change in soil K storage was estimated to be rather small (<0.10 kg/ha).

Summary and Conclusions

Water and nutrient balance studies carried out over a rice growing period of 70 days revealed the following:

- Of the average total water supply of 12.6 mm/d for the season, about 36% was supplied by irrigation. Of this about 52% was transferred by surface drainage, 36% was used as evapotranspiration and 12% was lost as deep percolation and seepage. For the majority of the time, evapotranspiration was more than pan evaporation.
- Substantial transfer of surface water can thus occur through percolation and seepage (average 0.7 mm/d and 0.8 mm/d respectively) and has the potential to contaminate groundwater. The percolation losses could possibly be reduced by early planting and thus reduce demand for irrigation, and high percolation losses during the off season.
- Approximate nutrient mass balances showed that approved plant uptake accounted for about 37%, 84% and 51% of the applied N, P and K respectively. The applied N not used by plants, either added to soil storage (27%) or was lost through various processes (36%). Denitrification appears to be the main process, as leaching, surface run-off and ammonia volatilisation accounted for relatively small proportions. As P is strongly sorbed on clayey soils its losses would primarily be through transfer of clay particles. Most of the K losses occurred through surface run-off.

Acknowledgments

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Field-Scale Nitrogen Leaching — Application of Model 'LEACHN'

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Abstract

The model LEACHN can be used to predict soil water transport, nitrogen transformations and nitrogen transport. This paper describes a study in which the LEACHN model was calibrated with field data collected at an experimental site in Perth, Western Australia, then subjected to a sensitivity analysis against several parameters. It was demonstrated how a calibrated model can be used for assessing the effect of modifying management in terms of applications of fertiliser and irrigation.

Model modifications needed to suit the tobacco agrosystem on the Kelantan Plain, Malaysia and prospects for applying the model to study physical conditions in this region are outlined.

THE BEHAVIOUR of N in the soil-plant-water system is very complex. It is dynamic and involves numerous interactions and transformations. Models are useful tools for integrating different processes involved in N transport in soil and can be used in forecasting how a system will behave without actually making measurements in the physical system (Tanji and Gupta, 1978). Clarke (1973) defines a mathematical model as 'A simplified representation of complex systems in which the behaviour of the system is represented by a set of equations, perhaps together with logical statements, expressing relations between variables and parameters'. In recent years development and application of models to predict soil water transport, N transformation, and N transport has increased tremendously. LEACHN is one such model which can be used to simulate field-scale N transformations and movement in the unsaturated zone of the soil profile. LEACHN is one of the five components of the model LEACHM (Leaching Estimation And Chemistry Model), developed by the Department of Agronomy at Cornell University, USA. The other components are: LEACHW (water only), LEACHP (pesticides), LEACHC (inorganic ions) and LEACHB (microbial transport). One particular application of LEACHN is the simulation of the leaching of nitrate (and ammonium) from the plant root zone, which is of much concern to the agriculturalists and environmentalists because once beyond the root zone, N is no longer available to plants and thereby has the potential to pollute the groundwater.

The objectives of this study were, firstly, to calibrate LEACHN with field data collected in an experimental site in Perth, Western Australia, secondly, to test the sensitivity of several parameters using the calibrated model, and thirdly, to plan a field data collection program on a tobacco farm in Kelantan for calibration and validation of the model.

Nitrogen in Agrosystems

Two primary sources of N applied in agriculture are chemical fertilisers (e.g. anhydrous ammonia, ammonium nitrate, urea, ammonium sulphate and ammonium phosphate) and organic products (e.g. manures, animal wastes, human wastes). Additional sources of N input include symbiotic fixation by plants, and irrigation and atmospheric input (Sharma 1993). N undergoes a wide variety of transformations in the soil. Following is a brief description of the processes.

Urea hydrolysis:	enzymatic hydrolysis of urea to ammonia.
Nitrification:	biological oxidation starting with ammonium-N through a series of intermediates to the nitrate form.

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Denitrification:	biological process involving the reduction of nitrate to gaseous end products (N_2 and N_2O), which eventually become part of the atmosphere.
Mineralisation:	conversion of organic forms of N to inorganic forms of NH_4 and NH_3. The process is performed by soil organisms that utilise nitrogenous organic substances as an energy source.
Immobilisation:	transformation of inorganic N compounds (NH_4, NH_3, NO_3, NO_2) into the organic state.
Volatilisation:	loss of ammonia to the atmosphere.
Leaching:	process of the downward movement (beyond the plant root zone) of dissolved constituents in soil solution.

Model Description

In LEACHM water transport calculations are carried out first, then the chemical transport is determined on the basis of these calculations. The soil profile is divided into layers of uniform thickness. Soil water flow is computed using the Richards equation, which is a second-order partial differential equation describing the flow of a fluid in a porous medium:

$$C(\theta)\frac{\partial h}{\partial t} = \frac{\partial}{\partial z}\left[K(\theta)\ \frac{\partial\ H(\theta,z)}{\partial z}\right]U(z,t) \quad (1)$$

Where θ is the volumetric soil water content, t is time, z is the vertical distance, $K(\theta)$ is the unsaturated hydraulic conductivity, H is the hydraulic head, U is a sink term representing water uptake by plant roots and $C(\theta)$ is the differential water capacity. Soil water potential and conductivity are calculated as a function of soil water content using Campbell's (1974) relationships. The model can also use soil textural characteristics and saturated hydraulic conductivity to estimate soil hydraulic properties in the absence of Campbell's parameters.

Solute movement is calculated using the advection-dispersion equation which includes adsorption, dispersion and advection terms. The general transport equation for solute movement is:

$$\frac{\partial c_L}{\partial t}(\theta + \rho K_d) = \frac{\partial}{\partial z}\left[(\theta)\ D(\theta,q)\ \frac{\partial c_L}{\partial z} - q\ c_L\right] \pm \Phi \ (2)$$

Where c_L is solute concentration in the liquid phase, *t* is time, θ is the volumetric water content, ρ is the bulk density of soil, K_d is the distribution coefficient, z is the vertical depth, (θ,q) is the apparent diffusion coefficient, q is the water flux and ϕ is the source or sink of solute.

The differential equations describing the flow of water, solute, and heat are solved by finite difference methods using a Crank-Nicholson implicit differencing scheme, which provides unconditional stability and relatively good accuracy.

The upper boundary can handle rainfall, irrigation, evaporation or zero flux. The lower boundary condition can be one of the following: zero flux, free drainage, lysimeter, fixed or fluctuating water table.

Potential evapotranspiration (PET) is determined from weekly pan evaporation using a pan factor. PET is allocated first to potential transpiration (PT) based on fractional crop cover and the rest is allocated to potential soil evaporation (PE). Actual transpiration and evaporation are calculated based on PE and PT and are limited by soil moisture content. Crop cover and root density may be constant or vary with time.

The model LEACHN includes N transformation processes such as mineralisation, nitrification, denitrification and surface volatilisation. The rate constants which control these processes are dependent on temperature and soil water. The root uptake of N is simulated using the Watts and Hanks (1978) approach.

The latest version of LEACHN includes a simpler water flow model based on Addiscott (1977). In this approach the soil segments are characterised by field capacity and a threshold between an 'immobile phase' and a 'mobile phase'. Water flow and solute transport occur only in the mobile phase; transport between mobile and immobile phases is in response to concentration gradients.

LEACHM cannot be used for:

- predicting runoff water quantity or quality;
- simulating the response of plants to soil or environmental changes;
- predicting crop yields;
- simulating the transport of immiscible liquids;
- predicting solute distributions in situations subject to two-or three-dimensional flux patterns (Hutson and Wagenet 1992).

Model Calibration

Description of the site

The model LEACHN was calibrated with field data collected in an experimental site located in the Perth metropolitan area, Western Australia. The area has a mediterranean climate with a rainy winter from May to October and a dry summer period from November to March. The average rainfall is around 800 mm/yr. During summer, irrigation is the primary source of water supply.

The experimental site was located at Noranda sporting complex and was developed in 1990. The soil is locally classified as Bassendean sand. It has a deep sandy profile of grey colour with 99% particles in the sand fraction. The soil has low water and nutrient retention capacity. The water table is about 3 m below the surface. The irrigation water supply is from bores which extract water from the underlying shallow aquifer. The water and nutrient fluxes passing below the root zone were measured by installing in-situ lysimeters (placed at 1.3 m from surface) in which leachate was collected, pumped out and measured for water quantity and chemical composition (Sharma et al. 1992). Amount and quality of irrigation water was also measured. Rainfall data was obtained from weather data published by the Bureau of Meteorology.

Input data

Some of the soil physical data used in the model are given in Table 1. Some of the data in the table have measured values. Hydraulic conductivity and retentivity parameters for Campbell's equation (AEV, BCAM) were calculated on the basis of soil water retention data obtained in the laboratory. The soil samples from the field were put under different tensions and their moisture content determined. Power regression analysis was performed on retention data to obtain the parameters. Most of the N transformation data used by the model developers were retained as no field data could be obtained. The model was run without consideration of the organic N and C pools. The reasons being no data on organic N and C pools for the simulation period were available, and it was assumed that slow N mineralisation in the top layers would ensure that most of the mineralised N would be taken up by the grass roots and thus leaching of mineralised N would be minimal.

Initial and boundary conditions

The soil profile modelled was 1.3 m in depth. At this depth the lower boundary was a lysimeter. The model simulates a lysimeter by using a combination of the constant potential and zero flux conditions. Water can drain from the bottom when that layer is saturated (or the moisture content is greater than field capacity, as in the capacity option of the model).

No data were available for the initial moisture content in the profile. As such, the profile was assumed to have a moisture content close to field capacity. Considering that the field was irrigated regularly during that period, this assumption is quite reasonable. The mineral N content in the soil layers were measured about a month before the beginning of the modelling period. The NO_3-N content in the top 60 cm of the profile was low (~ 1.5 kg/ha).

Parameter estimation

Calibration means that given a certain combination of parameters and boundary conditions, the model will

Table 1. Model input data (kg/dm^3).

Soil depth mm	Bulk density kg/dm^3	Saturated hydraulic conductivity (mm/d)	Roots[1] (relative)	AEV[1] KPa	BCAM[1]
0–100	1.5	3500	0.81	-1.25	1.72
100–200	1.5*	3750	0.06	-1.25	1.72
200–300	1.6	4000*	0.04	-1.25	1.72
300–400	1.6*	4000*	0.05	-1.25	1.72
400–500	1.55*	5000	0.03	-1.25	1.72
500–600	1.55	6000	0.00	-1.02	2.30
600–700	1.55	6500*	0.00	-1.02	2.30
700–800	1.55	6500*	0.00	-1.02	2.30
800–900	1.55	6500	0.00	-1.02	2.30
900–1000	1.55	6500	0.00	-1.05	1.97
1000–1100	1.55	6500	0.00	-1.05	1.97
1100–1200	1.55	6700*	0.00	-1.05	1.97
1200–1300	1.55	6700	0.00	-1.05	1.97

* Measured Values

[1] Based on measured soil moisture retention characteristics

AEV = Air entry value

BCAM = (Campbell 1974)

Roots relative = expressed as fraction of total predict measured values. In applying LEACHN to simulate the N movement in a field covered with grass, efforts were made to match output values of total drainage, total NO_3-N and NH_4-N loss in drainage, and concentration distribution of NO_3-N in leachate with measured values.

The simulation period was from the 28th November 1991 to 9th July 1992. The 1.3 m profile was divided into 13 segments of 100 mm each with a lysimeter boundary condition at the bottom. Grass covering the field was considered mature. Trial and error method of calibration was employed. At the beginning, parameters were adjusted to match the water flow components. It was also necessary to adjust initial water content to match drainage outflow. Once a reasonable match was obtained between model output and measured drainage, few other parameters were adjusted to match total loss of NO_3-N and NH_4-N loss in drainage water. In adjusting model parameters, the values chosen were reasonable. The first approximation was based mostly on published values. Many of the parameters were not adjusted at all. The values were those used by the developers of the model based on their experience.

Sensitivity analysis

A sensitivity analysis was performed on the calibrated LEACHN model (capacity version). It was observed that the most sensitive parameters were pan factor, annual potential N uptake by plants, and matric potential at field capacity. Sensitivity of different parameters were judged on the basis of their impact on total drainage amount, the amount of NO_3 and NH_4 lost in the leachate and volatilisation loss of NH_4. A slower nitrification rate (0.2/day in comparison with calibrated value of 0.4/day) only reduced NO_3 in leachate by 10%, but increased volatilisation by 45%. Doubling the K_d value for NH_4 resulted in a decrease in NH_4-N volatilisation (-33%) and also decreased NH_4-N loss in leachate. Doubling the volatilisation rate increased the amount of volatilisation (+81%) but reduced NO_3-N loss in leachate (-8%). Non inclusion of temperature effects had very little impact on model output. Changing the pan factor from 0.745 to 0.8 resulted in an increase in transpiration (+7%), reduction in drainage (-15%), reduction in NO_3-N loss in leachate (-9%), and an increase in plant NO_3-N uptake. When matric potential at field capacity was chosen as -10 KPa instead of the value used in calibration (-6 KPa), drainage increased by 14%. Increasing annual potential plant N uptake from 115 kg/ha to 172.5 kg/ha resulted in a sharp decrease in leachate NO_3-N loss (-40%) and an increase in NO_3-N uptake by plants (+32%). Details of the sensitivity analysis are provided in Table 2.

Table 2. Sensitivity analysis of model LEACHN, measured in terms of drainage, transpiration, leaching and plant uptake of NO_3-N and NH_4-N.

Drainage mm	Transpiration mm	NO_3-N (kg/ha)		NH_4-N (kg/ha) Volatilisation	Comments
		In leachate	Plant uptake		
380		30.2			Measured
378	736	30.2	38.5	4.2	Calibrated
378	736	32	38.5	3.1	Change nitrification rate from 0.4 to 0.8/day
378	736	27.1	38.3	6.1	Decrease nitrification rate from 0.4 to 0.2/day
378	736	31.2	40	2.8	Increase Kd for NH_4-N from 0.16 to 0.32 L/Kg
378	736	29.1	36.9	5.6	Decrease Kd for NH_4-N from 0.16 to 0.08 L/Kg
378	736	27.8	37.9	7.6	Increase volatilisation rate from 0.06 to 0.12/day
321	791	27.4	40.7	4.2	Change pan factor from 0.745 to 0.8
431	734	29.6	41.1	3.5	Change matric potential at field capacity from -6 Kpa to -10 Kpa
378	736	18.1	50.9	4.0	Increase annual potential N uptake by plants from 115 to 172.5 kg/ha

Table 3. Model output under different management options.

Drainage mm	Transpiration mm	NO_3-N (kg/ha)		NH_4-N (kg/ha) Volatilisation	Comments
		In Leachate	Plant Uptake		
380		30.2			Measured
378	736	30.2	38.5	4.2	Calibrated
378	736	59.0	42.4	6.6	Increase fertiliser application (NH_4-N) from 76 to 114 kg/ha (50% increase)
		(+95%)	(+10%)	(+57%)	
378	736	16.5	35.6	3.1	Decrease fertiliser application by 25% (from 76 to 57 kg/ha)
		(-45%)	(-8%)	(-26%)	
389	727	27.0	42.1	4.0	Once a week irrigation instead of twice a week
(+3%)		(-11%)	(+9%)	(-5%)	
378	736	27.9	41.1	4.0	5 fertiliser applications instead of 3 (total amount unchanged)
		(-8%)	(+7%)	(-5%)	

(Using LEACHN with capacity model)

Model output under different management options

The model was calibrated assuming twice weekly irrigation. If the same amount of irrigation was applied once a week, the model output shows that drainage would increase slightly (+3%), NO_3-N loss in drainage would decrease by 11% and plant uptake of NO_3-N would increase by 9% (Table 3). The calibrated model was applied with a 50% increase in fertiliser input. The output shows that NO_3-N loss in leachate would have increased 95% with only a slight increase (10%) in plant uptake. The plant, having extracted NO_3-N at almost potential rate could not increase uptake much further resulting in most of the N being lost in leachate. A 25% decrease in fertiliser application reduced NO_3-N loss in leachate by 45%, but plants had only 8% less NO_3-N uptake. Increasing the number of fertiliser applications from 3 to 5 (keeping the total amount unchanged) reduced NO_3-N loss in leachate by 8% and increased plant uptake by 7%.

Results and Discussion

The calibrated model output shows drainage of about 378 mm and 30.2 kg/ha of NO_3-N and 0.62 kg/ha of NH_4-N loss in leachate against the measured values of 378 mm, 30.2 kg/ha and 0.6 kg/ha respectively during the simulated period (Figs 1, 2, 3). At the end of the simulation period the model also predicts 1.47 kg/ha of NO_3-N in the soil profile which is also close to the measured value. From a management point of view such results would probably be acceptable, but the model output regarding NO_3-N concentration of leachate was not close to measured values. NO_3-N loss was much faster in reality. One possible reason is fertiliser was applied on 6 February 1992 and there was a heavy downpour (120 mm) on 9 February. Rainfall of such magnitude is very unusual in Perth at this time of the year. It is plausible that such heavy rainfall saturated the whole profile and created a preferential flow path bringing fertiliser N from top layers very quickly to the leachate. The measured values on 13 February 1992 shows a sudden increase of NO_3-N concentration of 23.3 mg/L from the previous week's concentration of 0.16 mg/L.

LEACHN has no capability to model macropore or preferential flow of water and solutes. Total NO_3-N loss in leachate in model simulation agrees fairly well with the measured value although the time taken for NO_3-N to travel to the bottom of the lysimeter is greater in the model simulation. Once N has moved below the top layer (100 mm) which contains 81% of the total root mass, most of it probably comes out in the drainage water. An event such as 120 mm rainfall in a very short duration can create a situation which is not possible to simulate in models like LEACHN.

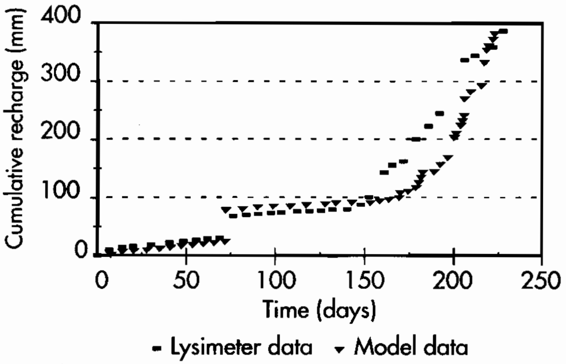

Figure 1. Measured and simulated cumulative recharge.

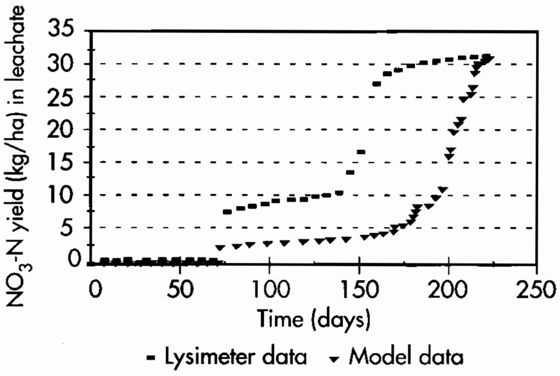

Figure 2. Measured and simulated cumulative NO_3-N loss in the drainage water.

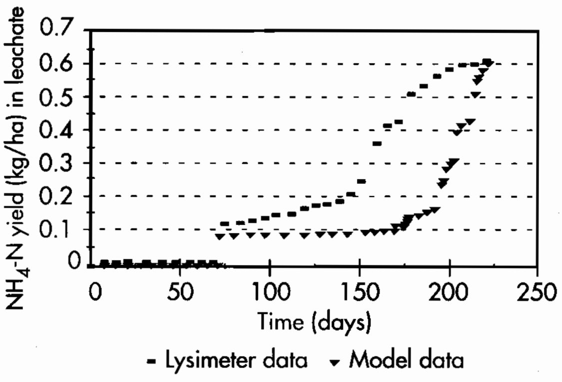

Figure 3. Measured and simulated NH_4-N loss in the drainage water.

The sensitivity analysis shows that the most sensitive parameters are pan factor, potential annual plant N uptake, and matric potential at field capacity. Although LEACHN requires many other parameters, most of them have very little impact on model output. It was also shown that management decisions can be made based on the model application, especially in the selection of fertiliser amount and irrigation scheduling.

Model Modification to Suit Tobacco Agrosystem

In order to use LEACHN for simulation of N dynamics in the tobacco agrosystem in Kelantan Plain, it is inevitable that a few modifications have to be made on the model. The proposed modifications are as follows:

a) The root distribution function

The default root distribution function in LEACHN was adopted from that of maize. This can be changed to that of tobacco using the empirical results of Ahmad (1992). The root dry weight distribution with respect to soil depth and plant age can be described by a characteristic function:

$$Y = A\left(\frac{1}{\sqrt{z}}\right)\exp\left(\frac{-C^{i2}}{4z}\right) \qquad (3)$$

Where A and C^i = root distribution parameters, z = distance (soil depth in cm), Y = root dry weight (g). The parameter A was found to correlate well with plant age with the regression equation

$$A = 2.58 * 10^{-3}\ t\ 1.78 \qquad (4)$$

By substituting equation (4) into equation (3) the root distribution function Y = f (z, t) is:

$$Y = 2.58 * 10^{-3}\ t^{1.78}\left(\frac{1}{\sqrt{z}}\right)\exp\left(\frac{-C^{i2}}{4z}\right) \qquad (5)$$

The C^i value was calibrated against the experimental root data and was estimated to be 0.9. The conditions for finite differencing in LEACHN has made it possible for the root distribution to be used to estimate average root length in each soil layer. The computed root dry weight of each segment can be converted to root length per unit volume of soil (L cm/dm^3) through another empirical function.

$$X = 0.747 + 5.051Y \qquad (6)$$

where X is the root wet weight and Y is the root dry weight. Equation (6) can be used to estimate the total fresh root weight of each soil layer and be converted to root length (L) by assuming that the roots were cylindrical and have an average radius of 0.05 mm and average specific root density of 1.1 g/cm^3.

b) The N-uptake process

The SINKN subroutine can be modified in terms of N uptake, taking into account the Nye/Warncke approach. N uptake by plant roots (U) is a function of NH_4 and NO_3 concentrations and root length density (L) and can be calculated from

$$U = 2 \pi r c \alpha L \Delta t \qquad (7)$$

where r is the mean root radius (m), c is the nutrient concentration in solution (mg/m^3), α is the root uptake coefficient (m/day), L represents the root length density (m/dm^3) as calculated above and t is the time (day). The relationship between α and NH_4 and NO_3 can be expressed by (Warncke and Barber 1973)

$$\log \alpha NH_4 = -6.81 - 0.63 \log [NH_4] - 116.8 [NO_3] \qquad (8)$$

$$\log \alpha NH_3 = -6.80 - 0.61 \log [NH_3] - 127.1 [NO_4] \qquad (9)$$

where the concentrations are in mol/L.

c) The water retention characteristics

At present LEACHN uses Campbell's water retention and conductivity equations. LEACHN also provides a choice of water retention regression models for predicting water retention parameters from particle size distribution, bulk density and organic matter content. The equations were developed for regional data sets, and may not be applicable to soils which may have developed on different parent materials or under different climatic conditions (Hutson and Wagenet 1992). Efforts will be made to include other retentivity and conductivity functions in LEACHN (see Van Genuchten 1980) so that a user has the option to choose the function that suits the local conditions.

Application of LEACHN at a Tobacco Farm in Kelantan, Malaysia

A tobacco farm in Kelantan (T-07) was selected for intensive data collection. The purpose was to gather the necessary data for LEACHN input. Lysimeters were placed at a depth of 75 cm. Data were collected on soil physical properties, for example, saturated hydraulic conductivity, bulk density, and soil moisture retention characteristics. The amount and rate of fertiliser application, the amount and timing of irrigation including irrigation water quality and rainfall data were also collected. Plant N uptake was also measured. Unfortunately due to frequent flooding the lysimeter data could not be used. Consequently, a nutrient and water balance calculation could not be made. Efforts will be made this coming growing season (December 94–April 95) to collect the necessary data for LEACHN calibration. To safeguard against high water table, data will be collected for soil NH_4 and NO_3 at different depths. Porous cups will also be used to gather soil water NH_4 and NO_3 concentrations. LEACHN output includes the amount of NH_4 and NO_3 both in soil and in soil water at different layers. If lysimeter data are not available for the whole growing season then soil and soil water NH_4 and NO_3 can be used for calibration purposes.

Conclusion

The model LEACHN has been used with field data for simulating nitrate leaching in an experimental site in Western Australia. Based on the sensitivity analysis and simulation results the following conclusions can be made:

- Only few (pan factor, annual potential N uptake by plants and matric potential at field capacity) of the parameters have strong influence on model output.
- Slower nitrification rate, higher volatilisation rate, higher N-uptake by plants resulted in reduced NO_3 loss in leachate.
- The model can be used as a management tool in decision making, for example, amount of fertiliser input and frequency of irrigation and fertiliser application.
- By increasing the number of fertiliser applications (keeping the total amount unchanged) the NO_3 loss can be reduced and plant uptake increased.
- The calibration process highlighted the inability of the model to simulate macropore flow.
- Suggestions have been put forward to modify the models to suit local conditions in terms of root distribution, N-uptake process and simplify expression of soil water retention characteristics.

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Piezometer Grid Studies to Assess Pollution Sources at Tobacco and Paddy Farms, Kelantan, Malaysia

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Abstract

As an adjunct to the broadscale study of the impact of agricultural practices on the environment in rice and tobacco areas of the Kelantan Plain, Malaysia, groundwater grid studies were designed to quantify and evaluate the short-term dynamics of selected pollutants in the groundwater. This paper describes the results obtained from two representative areas of rice and tobacco. Piezometer studies revealed extensive pollution of groundwater by inorganics, organic pesticides and microbial contamination. There was evidence of contribution of inorganic pollution (especially nitrate) from point sources on the farms. Simplified models were applied to estimate travel times for leaching of selected pollutants through soil profiles (under rice and tobacco cultivation) and also for their transport within the aquifers.

FOLLOWING a broadscale survey in the rice and tobacco growing areas of the Kelantan Plain, Malaysia, it was revealed that drinking water wells in these areas exhibited widespread contamination. The major inorganic pollutants in the tobacco area were nitrate, and total dissolved solids (TDS) while in the rice area NH_4 was the main pollutant (Ahmad et al. these proceedings). Elevated levels of natural Fe were observed in both agrosystems. Chlorinated pesticides were also detected in both agrosystems. Carbofuran was detected in the tobacco area, while 2,4-D and paraquat were detected in the rice area (Cheah et al. these proceedings). Microbial contamination was also found to be widespread in both agrosystems, especially during the rainy season when flooding occurred (Lum these proceedings). In addition to agrochemicals used on the farms, other possible sources of pollutants may include point sources such as household toilets, manure storage sheds, and animal excreta.

The groundwater grid studies were designed to quantify and evaluate the short-term dynamics of selected pollutants in the groundwater beneath typical tobacco and rice farms of the Kelantan Plain. The aim was to gain a better understanding of the sources of pollution in the drinking water wells, to examine the interrelationships of various factors affecting the level of pollution, and to make estimates of travel times of selected pollutants.

Methods

Two experimental sites were selected in tobacco and rice growing areas, encompassing representative tobacco and rice farm households. These sites were incorporated as observation points in the broadscale survey, and exhibited high pollution levels.

Experimental Design and Measurements

Tobacco site

The study area (Fig. 1) is located near the eastern coast and includes T-07 farm (a sampling point in the broadscale survey) and some areas from surrounding farms, which are typical to the area. The landscape has a low gradient and is mostly low lying. The soils are sandy bris (beach ridges interspersed with swales).

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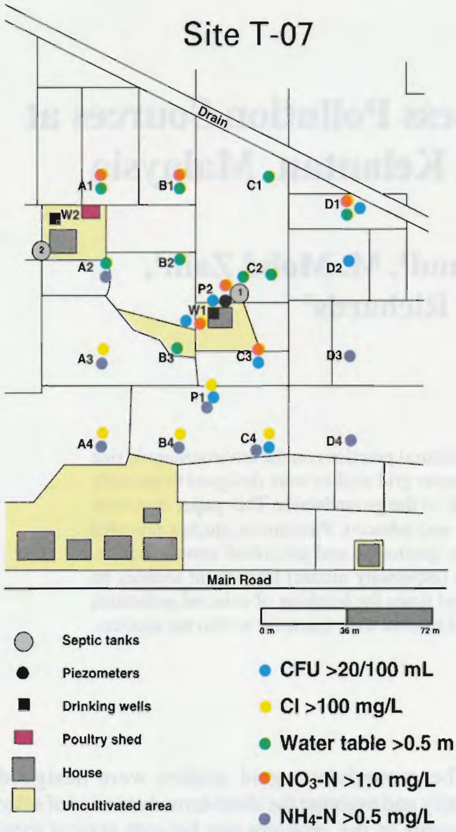

Figure 1. Map showing the location of piezometers in relation to drinking wells at the tobacco experimental site T-07. Also shown are locations where mean values of measured pollutants (NH_4-N, NO_3-N, Cl, CFU) and water table depth exceeded an arbitrary value.

For convenience of cultivation and other farming operations, the area was divided by farmers into identifiable plots of uneven size (marked with letters in Fig. 1). An earthen drain was installed to allow groundwater discharge to minimise the impact of flooding.

Sixteen piezometers were installed at a depth of 2 m on a 36 m grid pattern. Two additional piezometers (P1, P2) were installed near the household. With the inclusion of the T-07 drinking water well, a total of 19 locations were sampled. Water samples from the 19 locations were collected weekly for 13 weeks over a tobacco growing period (11/01/93 to 26/04/93).

At each sampling the depth of the water table was measured, as well as temperature, pH, redox potential (Eh) and electrical conductivity (EC) of the water sample. The samples were analysed for various inorganic (NO_3, NH_4, TDS, PO_4, K, Fe, Cl) organic (organochlorines, methamidophos) and microbiological (colony-forming units — CFU) levels.

All the agronomic operations were recorded. Irrigation water was derived from the underlying shallow aquifer. There was a large variability in fertiliser application among individual plots. Ranges included 14–108 kg/ha N, 25–158 kg/ha P, 37–211 kg/ha K. Methamidophos 48.4% 'Tamaron special' diluted at a rate of 2 ml/L, was applied at a rate of 1.2 L/ha–2.9 L/ha.

Rice site

The study area includes sampling site R-21 which is located in Kg. Pohon Tajung, 10 km northwest of Pasir Mas Town (Fig. 2). The site elevation is 16 m above sea level. The soil is of riverine origin, consisting of Batu Hitam, Kg. Tepus and Sg. Amin series. This soil is characterised by clay to loam texture, with the presence of stratifications, and it is well to poorly drained.

Sixteen piezometers were installed in a rectangular grid system, with some irregularities (Fig. 2). The household was taken as the centre of the observation unit. An additional piezometer was installed between the household toilet and drinking well.

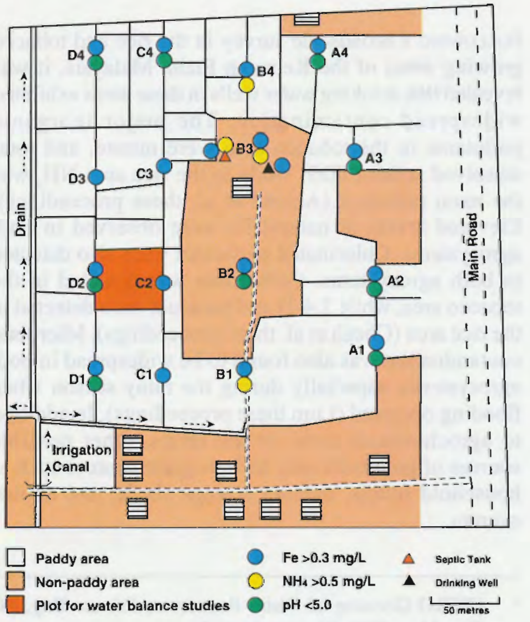

Figure 2. Map showing the location of piezometer in relation to drinking well at the paddy experiment site R-21. Also shown are locations where mean values of selected parameters (pH, NH_4-N, Fe) exceeded an arbitrary value.

Water samples from piezometers were collected weekly or fortnightly using a peristaltic pump with an in-line filtration system. Field measurements included EC, Eh, pH, temperature and depth of water table. Water samples were analysed for various inorganic (NH_4, PO_4, Fe) organic (endosulphan and lindane) and microbiological (CFU) levels.

Results and Discussion

Tobacco

Pollution levels

Inorganic: Means and standard deviations of NO_3-N, NH_4-N, PO_4-P, K, Fe, pH, Eh, EC and Cl measured over 13 observation periods, using 19 piezometers are presented in Appendix 1. Overall the variation associated with the mean was much higher for means averaged over space than those averaged over time. Examination of spatially-averaged means showed that NH_4, K, PO_4 and Fe exceeded the Malaysian standard for drinking water on all occasions, while NO_3, Cl and pH did not exceed the standard on any occasion. However, based on time-averaged means, it was found that Fe exceeded the limit at all locations, NH_4-N and K at nine locations (47%) and NO_3-N and Cl at two locations (10%).

At four locations (A1, B1, P1 and W1) NO_3-N levels exceeded the drinking water standard. The frequency of such occurrence was much higher at W1 and P1 (>25%) which were located close to the toilet, and at A1 and B1 (19%) which were located close to the poultry shed.

It was found that Eh was strongly correlated with NO_3, and negatively with PO_4-P, NH_4 and Fe. Relationships of NH_4 with pH, and water table depth showed that high NH_4 levels occurred when pH >7 and water table depth was within the top 30 cm.

High water tables are associated with low Eh (Appendix 2) and therefore, the rate of mineralisation of NH_4 may have been inhibited due to anaerobic conditions (Ahmad et al. these proceedings). A direct relationship between NO_3 with Eh, as shown in Figure 3 is consistent with this.

A strong dependence of PO_4-P levels on Fe and pH is shown in Figure 4. PO_4-P levels were >1 mg/L when pH <7 and Fe >10 mg/L. As in the case of NH_4, PO_4-P levels were higher when water table depth was <30 cm (Eh low, anaerobic condition) and the reverse happened under high pH and Eh conditions.

In general the Eh values decreased with an increase in the water table, reflecting anaerobic conditions. The Eh is determined by the population of anaerobes and the amount of organic matter. Under low Eh (high water table) conditions, NH_4 would remain unoxidised, and NO_3 would be denitrified to N_2O and N_2. The ferric iron (Fe^{3+}) would be reduced to ferrous (Fe^{2+}) releasing adsorbed PO_4-P into the solution. As water recedes, aerobic conditions will occur, resulting in mineralisation of NH_4 to NO_3. Fe^{2+} would be re-oxidised to Fe^{3+} hydroxides, and P would be readsorbed. If the water table rises again, Eh would drop and anaerobic reactions would start all over again, resulting in increased Fe and PO_4-P and reduced NO_3. The observed variations in NO_3 were found to be closely related to such variations in water table depth and Eh.

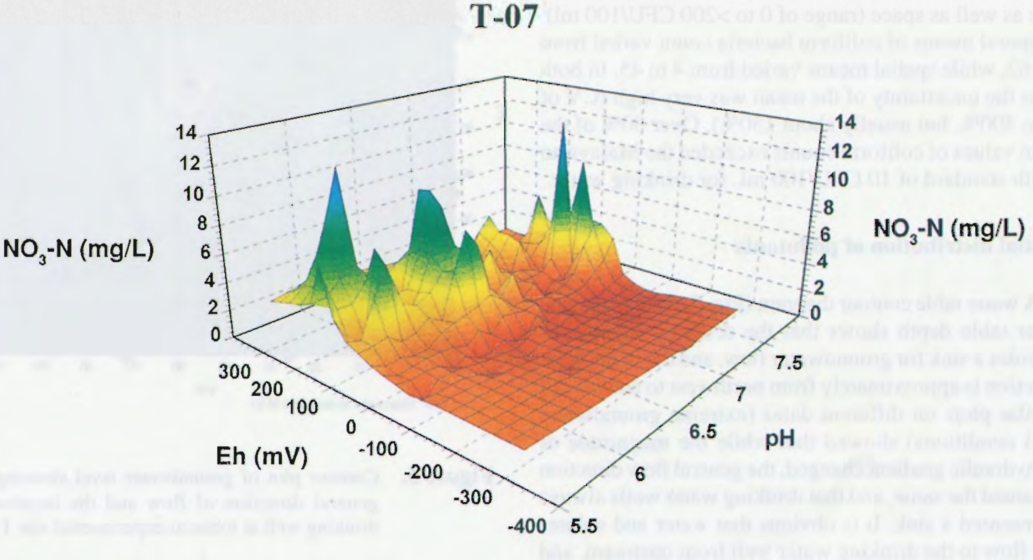

Figure 3. 3-D plot of average NO_3-N, Eh and pH values at tobacco experimental site T-07.

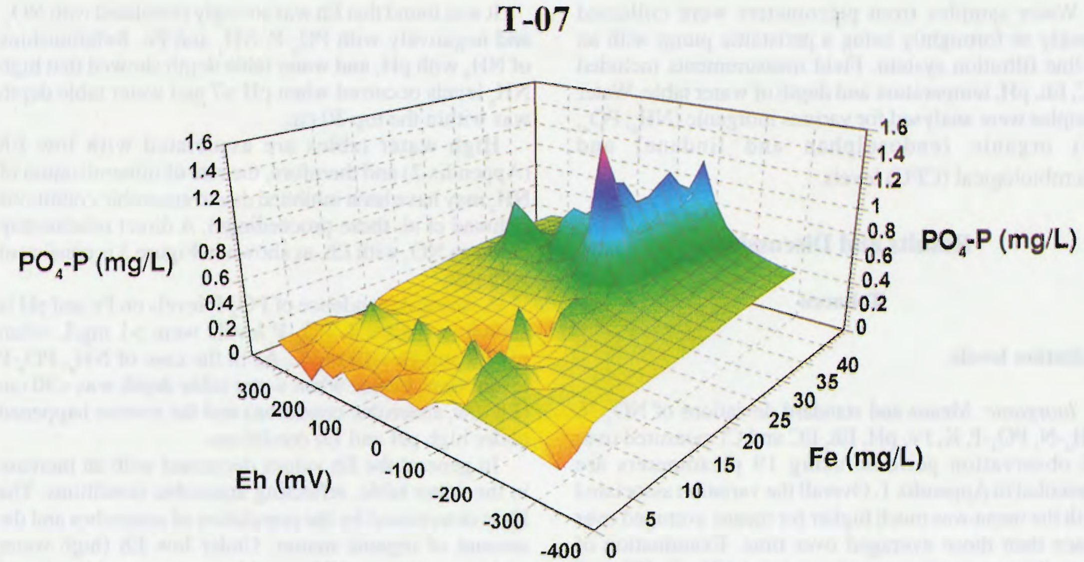

Figure 4. 3-D plot of average PO$_4$-P, Eh and Fe values at tobacco experimental site T-07.

Organic: All the groundwater samples were analysed for methamidophos, currently being used in tobacco growing. Methamidophos was not detected in any of the samples.

Microbial: Total coliform counts at various locations and sampling times are presented in Appendix 1 along with their spatial and temporal means and standard deviations. Microbial activity varied dramatically with time as well as space (range of 0 to >200 CFU/100 ml). Temporal means of coliform bacteria count varied from 3 to 63, while spatial means varied from 4 to 45. In both cases the uncertainty of the mean was very high (CV of up to 300%, but usually about 150%). Over 80% of the mean values of coliform counts exceeded the Malaysian health standard of 10 CFU/100 mL for drinking water.

Spatial distribution of pollutants

A water table contour diagram (Fig. 5) for the average water table depth shows that the drinking water well provides a sink for groundwater flow, and the water flow direction is approximately from northwest to southwest. Similar plots on different dates (extreme groundwater level conditions) showed that while the magnitude of the hydraulic gradient changed, the general flow direction remained the same, and that drinking water wells always represented a sink. It is obvious that water and solutes will flow to the drinking water well from upstream, and flow out to the downstream side.

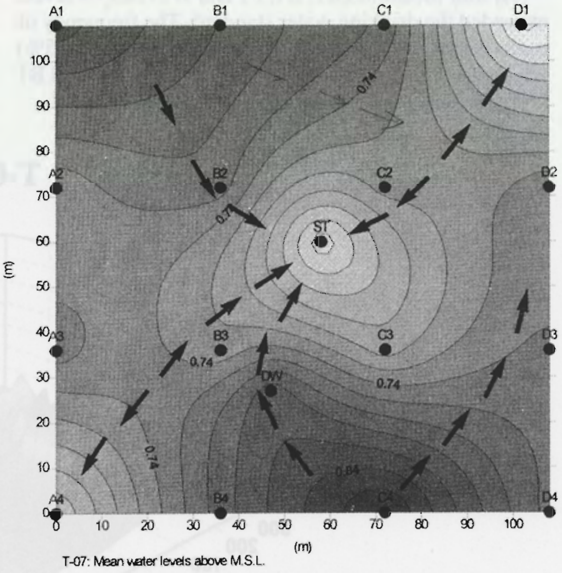

Figure 5. Contour plot of groundwater level showing the general direction of flow and the location of drinking well at tobacco experimental site T-07.

In addition to the spatial distribution of elevated levels of pollutants (above an arbitrary level) the contour diagram of NO$_3$-N concentration for the site (Fig. 6a) clearly shows that high NO$_3$-N is associated with proximity to the toilet (P1,W1) and the poultry shed (A1, B1). Several piezometers (A2, A3, A4, B4, C4, D4, P1) had mean NH$_4$-N concentrations >1 mg/L. Consistently high NH$_4$-N concentrations along with high levels of NO$_3$-N at P1 were associated with an organic source (toilet) as the site is on reasonably high ground (low water table depth, high Eh). However, the NH$_4$-N concentration in the groundwater well, although exceeding the health standard, is not too excessive.

There was some correspondence of NO$_3$-N concentration with the distance from the contamination source indicating that these point sources enhance enrichment of NO$_3$-N in groundwater, adding to that contributed by the fertilisers. Such a relationship for NH$_4$ with distance from point source was not obvious from the data.

High bacterial counts (Fig. 6b) were correlated with the location of the household, the drinking well and locations associated close to the septic tank.

Rice

Pollution levels/interrelationships

Inorganic: Means and standard deviations of the inorganic chemicals, measured over 14 observation periods using 18 piezometers are presented in Appendix 2. In all cases, water does not meet the drinking standards with respect to Fe and pH. With respect to NH$_4$-N concentrations, some 20% of locations exceeded the standard limit. Overall, pollution was rather small, with negligible amounts of NO$_3$.

Spatial distribution of pollutants

Water table levels and the contour diagram (Fig. 7) for the average water table depth for the area show that the groundwater sink coincides with the location of the drinking well, and that the general direction of flow is from southwest to northeast. Groundwater plots at other extreme conditions depicted similar patterns, although the magnitude was different. It is obvious that groundwater flowing from the rice field will pass through the well and may influence its quality.

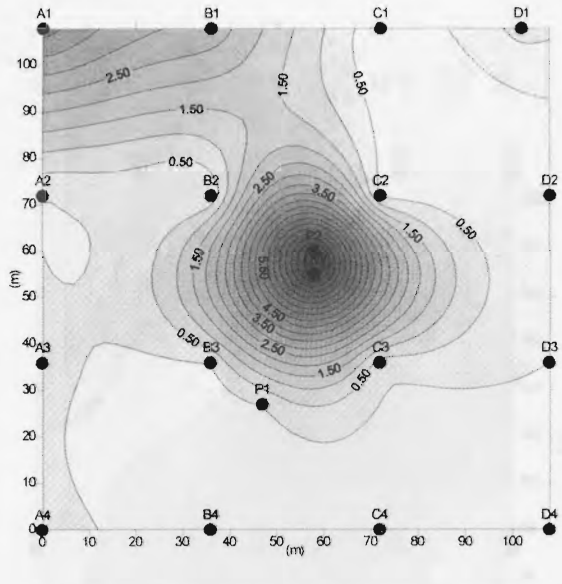

Figure 6a. Contour plot of NO$_3$-N concentration showing areas of elevated pollution at tobacco experimental site T-07.

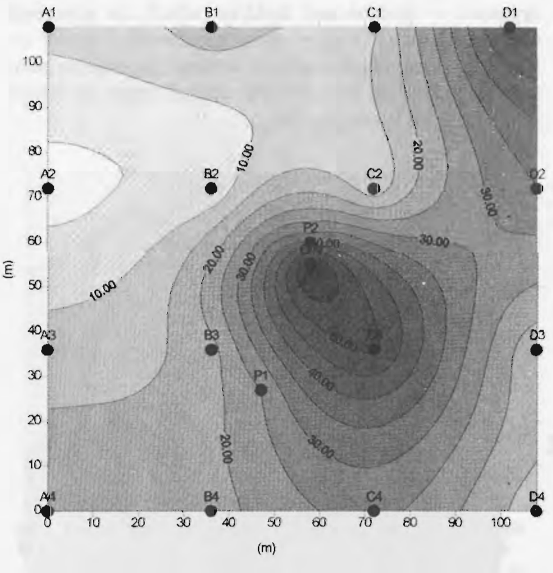

Figure 6b. Contour plot of microbial counts (CFU) showing areas of elevated pollution at tobacco experimental site T-07.

Spatial distribution of elevated levels of NH_4-N (above an arbitrary level) (Fig. 8) indicate that elevated NH_4-N levels are closely associated with the household, especially the location of the toilet. One location downstream from the household (B4) also showed elevated levels. A similar elevated level upstream (B1) may not be easily explained.

Estimates of travel times of pollutants

As discussed, because of groundwater flow gradients from the surrounding farm areas to the drinking well, pollutants move with groundwater. Therefore, estimates were made for the travel times for the following:

- time required to travel from the soil surface to the groundwater table, assuming a water table depth of 1 m, and
- time required to travel from the groundwater table at the edge of the farm to the drinking well of the household, at a distance of 10 m.

The travel of a pollutant through soil depends not only on the water retention and flow characteristics of the medium, but also on the decay, transformation, and absorption characteristics of the pollutant within the medium. Estimates were made for three pollutants, two organics — lindane and dieldrin (which are adsorbed and are subject to decay) and a conservative tracer — chloride. The latter is used as a standard for comparison, and will reflect comparable travel times of other pollutants, for example, NO_3.

It must be emphasised that the estimates were made under simplified assumed boundary conditions, and should be treated as a guide only. They should provide good relative travel time estimates for the given soil provided assumed conditions are met.

Pollutant travel from soil surface to groundwater table

These times for travel through 1 m soil profile depth were computed in two different ways. The first method was based on an assumed piston-type flow, and dispersion and decay were negligible. The residence time (*T*) for 1 m depth (*d*) was computed from:

$$\frac{d}{T} = \frac{R}{\rho K_D} \qquad (1)$$

where *R* is the recharge rate, ρ is the bulk density and K_D is the distribution coefficient for the pollutant in the medium, *d/T* is the mobility or travel rate.

The second method is based on the analytical solution of the 1-D advective-dispersion partial differential equation:

$$R \frac{\partial c}{\partial t} = D \frac{\partial^2 c}{\partial x^2} - v \frac{\partial c}{\partial t} - \mu c \qquad (2)$$

Figure 7. Contour plot of groundwater level, showing the general direction of flow, and the location of drinking well at R-21, a sampling point in the broadscale survey.

Figure 8. Contour plots of NH_4-N concentration at sampling point R-21.

for the following initial and boundary conditions:

$$c\,(x, 0) = c_i \qquad (3)$$

$$\left(-D\frac{\partial c}{\partial x} + vc\right)\Bigg|_{x=0} = \begin{cases} vc_0 & 0 < t < t_0 \\ 0 & t > t_0 \end{cases} \qquad (4)$$

$$\frac{\partial c}{\partial x}(L, t) = 0 \qquad (5)$$

where

c = solution concentration
R = Retardation factor = $1 + \rho K_d/\theta$
ρ = bulk density
K_d = distribution constant
θ = water content
D = dispersion coefficient
V = q/θ = pore water velocity
q = volumetric flux
μ = rate constants for decay
t_0 = duration of the pulse

Application

Computations of mobility (≈*d/T*) and residence time (*T*) for the tobacco conditions made were: $\rho = 1.5$ kg/m^3, depth 1 m, under two recharge (*R*) conditions (1 m/yr, and 0.2 m/yr).

Half life values of 1000 days for lindane and 5000 days for dieldrin, and $K_D = 14$ L/kg and $K_D = 117$ L/kg respectively were taken (Gerritse et al. 1991). Based on measured soil hydraulic properties ($K_s = 1.9$ m/day, $\theta_s = 0.43$) the corresponding soil water contents for the two recharge conditions of 1 m/yr and 0.2 m/yr were computed to be 0.14 and 0.11 for the typical tobacco sand. A single application of 10 kg/ha was assumed, and the initial concentration of pollutant in the profile was assumed 0 mg/L. Dispersivity (*D*) value of 50 cm^2/day was taken based on earlier work (Sharma et al. 1991).

The computed values of mobility (*d/T*) and residence time (*T*) for tobacco are presented in Table 1.

Table 1. Computed mobility (d/T) and residence times (T) for lindane and dieldrin for tobacco sandy soil using equation (1), with two recharge rates.

Pollutant	**Lindane**		**Dieldrin**	
Recharge (m/yr)	*d/T* (mm/yr)	*T* (yrs)	*d/T* (mm/yr)	*T* (yrs)
1	47.6	21	5.7	175
0.2	9.5	105	1.1	878

A commercially available computer program (ONE-D) was used, which solves equation (1) analytically (Van Genuchten and Alves 1982). The resultant concentration of pollutants (lindane, dieldrin, chloride) at 1 m depth under two recharge conditions as a function of time (after application) are presented in Table 2.

Table 2. Resultant pollutant concentration (mg/L) at 1 m depth as a function of time (after application, t') for two assumed recharge rates.

Lindane				**Dieldrin**				**Chloride**			
t'		R (m/yr)		t'		R (m/yr)		t'		R (m/yr)	
(days)	(years)	1	0.2	(days)	(years)	1	0.2	(days)	(years)	1	0.2
500	(1.4)	0	0	10 000	(28)	0	0	5		0.003	0.002
1000	(2.8)	0.0002	0	15 000	(42)	0.0001	0	25		4.66	2.42
2500	(7)	0.033	0.0006	25 000	(70)	0.0001	0	50		4.49	4.81
5000	(14)	0.011	0.0005	35 000	(97)	0	0	100		1.44	4.79
10 000	(28)	0	0					250	(0.7)	0.038	2.65
								1000	(2.8)	0	0.32

Discussion

Obviously pollutants will arrive in groundwater much earlier under high recharge. For example, in tobacco it will take 21 yrs and 105 yrs for lindane to travel 1 m depth under recharge rates of 1 m/yr and 0.2 m/yr respectively. The corresponding travel times for dieldrin are 175 yrs and 878 yrs respectively. These are the worst scenarios. A more realistic solution (Table 2) shows that the arrival times under 1m/yr recharge for lindane and dieldrin are much shorter, being about 1000–2000 days (~5.5 yrs) and ~ 15 000 days (~ 42 yrs) respectively. Such arrival time for chloride was much shorter (50–100 days).

Pollutant travel from field to the drinking well

This involved the use of an analytical solution of a partial differential equation (PDE) describing three-dimensional advective-dispersive transport of a non-conservative solute. It uses a condition when the solute is injected instantaneously from a point source into the aquifer, with the following initial and boundary conditions:

$$c\,(x, y, z, o) = 0 \qquad (6)$$

$$Q_c\,(x, y, z, o) = M/\mu \qquad (7)$$

$$c\,(\pm\infty, \pm\infty, \pm\infty, t) = 0 \qquad (8)$$

Where $M = Q_w\ C_0 dt$ is the total mass instantaneously released at the source. The following analytical solution given by Hunt (1963) was used:

$$c\,(x, y, z, t) = \frac{M}{8n\sqrt{\Pi^3\, t^3\, D_x\, D_y\, D_z / R}} \qquad (9)$$

$$\exp\left[-\lambda t - \frac{1}{4t}\left(\frac{(x - V t / R)^2}{D_x / R} + \frac{y^2}{D_y / R} + \frac{x^2}{D_z / R}\right)\right.$$

The commercially available computer program SLUG-3D, which is one form of model SOLUTE (Belijin and Heidje 1993) was used to solve the above equation both for tobacco and rice areas using assumed or estimated transport properties of the respective soil/aquifer systems. In both cases a solute slug was assumed to be applied at 10 m upstream of the drinking well. A single injection of 5 kg of solute was made, when the initial concentration was zero. For both conditions the common properties were: $\rho = 1.5\ \mathrm{Kg/m^3}$, $\theta_s = 0.43$, $D_L = 10$ m, $D_z = D_y = 1$ m, $dh/dx = 0.02$. For tobacco $K_s = 1.9$ m/d and for rice $K_s = 0.1$ m/d.

Resultant concentrations of pollutant (lindane, dieldrin, chloride) in the drinking well as a function of time (after application) for the tobacco and rice conditions are presented in Table 3.

Discussion

As expected, pollutants will arrive much more quickly through the tobacco sandy aquifer than the clayey aquifer of rice. In all cases the rate of movement of chloride is much faster than lindane, which in turn is much faster than dieldrin. For example, in a tobacco sandy aquifer a conservative tracer (e.g. Cl) will arrive within five days, while lindane will take some 500 days (~1.5 yrs) and dieldrin will take some 2500–5000 days (~10 yrs). The arrival times of the peak for the three pollutants are approximately 20 days, 750 days (~ 2 yrs) and 5000 days

Table 3. Resultant concentration of pollutant (c mg/L) in the drinking well as a function of time (days after application, t') when a single application was made at 10 m away from the drinking well in the upstream flow line in the field. The estimates are for tobacco and paddy aquifers.

Tobacco						**Paddy**					
Lindane		**Dieldrin**		**Chloride**		**Lindane**		**Dieldrin**		**Chlorine**	
t'	c	t'	c	t'	c	t'	c	t'	c	t'	c
100	0.004	1000	0.004	5	1.57	500	0	500	0	5	0
500	2.13	2500	0.089	10	9.41	1000	0	10 000	0	25	0
750	2.49	5000	0.227	20	13.48	2500	0.0001	30 000	0.0003	50	0.025
1000	2.16	10 000	0.125	50	7.48	5000	0.0012	50 000	0.0001	100	1.96
2500	0.42	20 000	0.019	100	3.15	7500	0.0007			250	12.28
5000	0.32	30 000	0.003	1000	0.02	10 000	0.0002			1000	7.06
10 000	0.003									5000	0.61

(~14 yrs) respectively. It must be emphasised that once the sorbed pollutants enter the aquifer, it takes a very long time before they disappear (or reduce to very low concentrations). Table 3 shows the times required to clean up the aquifer (without further applications) with respect to Cl, lindane and dieldrin will be about 1000 (~ 2.7 yrs) 10 000 (27 yrs) and >30 000 days (~ 83 yrs) respectively. The results were very similar for rice, except that much longer times are required. For example, peak arrival times for Cl, lindane and dieldrin will be approximately 250, 5000, 30 000 days respectively. The corresponding clean up times will be >5000, >10 000, >50 000 days, respectively.

Summary and Conclusions

Two experimental sites were selected in tobacco and rice areas, encompassing representative farm household drinking water wells. These sites were installed with a grid of piezometers, and their groundwater quality was measured and compared to the drinking water well over a crop season. Based on the examination of these data, and application of some simplified models in compiling the travel times of selected pollutants, the following conclusions were derived:

- The inorganic pollutants of concern in tobacco were: NO_3, NH_4, PO_4, K, Fe and Cl and their average values exceed drinking water standards in at least 10% of cases. Groundwater in rice areas did not meet the drinking water standard with respect to Fe and pH in most cases, and NH_4 in about 20% of cases.
- Piezometers in tobacco areas showed elevated levels of NO_3-N and NH_4-N especially close to point sources (toilet, poultry shed). Rice-growing areas with elevated NH_4-N levels were also close to the toilet.
- In a tobacco growing area P concentrations were closely related to Fe and Eh and pH, while those of NO_3 and NH_4, varied with respect to Eh and pH. Such relationships were not so obvious under rice.
- The bacterial pollution was very high, exceeding drinking water standards in 80% of cases. The drinking wells and locations close to septic tanks had the most consistently high bacterial count.
- Model applications demonstrated that in tobacco areas, travel times for Cl, lindane and dieldrin to leach through 1 m profile and reach groundwaters were in the order of 0.3, 5.5 and 40 yrs respectively. Furthermore, the estimated times for their travel 10 m from the field to the drinking well were even shorter (i.e. 0.1, 1.5 and 10 yrs respectively). In the rice area the corresponding travel times are expected to be higher.
- Once the sorbed chemicals (lindane, dieldrin) enter the aquifer, it will take a very long time before they disappear (~ 30 and 80 yrs in tobacco and rice areas respectively).

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Appendix 1. Spatial and temporal mean values (with standard deviation) of various inorganic chemicals measured at T-O7.

Sampling date	pH		Eh (mV)		WT (m)		NH_4-N (mg/L)		NO_3-N (mg/L)		PO_4-P (mg/L)		Fe (mg/L)		Cl (mg/L)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
11/10/93	7.1	0.3	-55	35	0.8	0.1	1.20	1.3	0.52	1.98	0.24	0.22	3.91	3.65	90	122
17/01/93	7.0	0.3	-68	37	0.7	0.1	1.09	1.2	0.52	1.80	0.31	0.33	4.72	4.55	86	109
24/01/93	7.0	0.4	-45	26	0.6	0.1	1.03	1.1	0.53	1.79	0.30	0.35	4.41	4.21	86	107
31/01/93	6.9	0.5	-54	87	1.0	0.1	1.10	1.1	2.02	4.24	0.32	0.41	4.72	5.63	113	177
08/02/93	7.0	0.4	-35	87	0.8	0.1	0.97	1.0	1.37	3.15	0.36	0.42	5.68	6.25	91	119
15/02/93	7.0	0.4	101	135	0.6	0.1	0.93	1.0	1.60	3.61	0.32	0.36	5.75	7.18	92	119
23/02/93	7.1	0.4	-53	57	0.4	0.1	0.99	1.0	1.25	3.14	0.29	0.31	5.82	8.21	99	133
01/03/93	7.1	0.4	-72	67	0.4	0.1	0.66	0.7	1.43	4.09	0.37	0.33	4.69	7.16	104	132
08/03/93	7.1	0.5	-63	73	0.5	0.1	0.76	0.8	1.46	4.10	0.38	0.35	5.72	9.63	103	132
15/03/93	6.5	0.5	-27	78	1.0	0.2	0.67	0.7	2.03	3.83	0.36	0.36	5.14	8.77	95	130
31/03/93	6.6	0.5	-13	67	0.7	0.1	0.65	0.7	1.46	2.83	0.34	0.34	5.93	7.91	96	140
17/04/93	6.6	0.5	94	40	1.1	0.1			2.13	3.71	0.26	0.33	3.80	6.41	94	141
26/04/93	6.7	0.5	127	63	0.8	0.1			2.99	6.00	0.26	0.33	4.96	6.97	86	123

Sampling point	pH		Eh (mV)		WT (m)		NH_4-N (mg/L)		NO_3-N (mg/L)		PO_4-P (mg/L)		Fe (mg/L)		Cl (mg/L)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
A1	6.9	0.3	0	80	0.9	0.4	0.16	0.1	4.26	5.82	0.04	0.03	1.43	0.91	15	3
A2	6.8	0.2	-14	64	0.7	0.3	2.69	0.7	0.00	0.00	0.14	0.06	4.75	0.94	61	15
A3	6.4	0.3	-34	76	0.8	0.2	1.59	0.9	0.17	0.18	1.15	0.37	27.44	9.62	262	55
A4	7.4	0.2	-25	79	0.6	0.2	2.64	1.1	0.13	0.32	0.26	0.11	1.16	0.65	354	117
B1	6.7	0.3	4	92	0.8	0.3	0.22	0.1	2.87	3.24	0.12	0.12	2.70	1.00	23	4
B2	6.6	0.2	-3	75	0.8	0.3	0.18	0.1	0.00	0.00	0.59	0.16	5.02	1.68	20	5
B3	6.3	0.3	-15	87	0.7	0.2	0.17	0.1	0.01	0.02	0.06	0.03	11.23	3.14	20	10
B4	7.4	0.2	-49	83	0.8	0.2	1.69	0.6	0.00	0.01	0.65	0.17	1.57	0.62	200	31
C1	7.1	0.3	-59	145	0.8	0.2	0.23	0.1	0.08	0.15	0.06	0.04	6.74	2.47	20	5
C2	6.5	0.3	8	96	0.7	0.3	0.19	0.1	0.30	0.61	0.38	0.22	3.43	1.47	20	5
C3	6.5	0.5	-63	53	0.7	0.2	0.13	0.1	0.22	0.40	0.09	0.05	10.61	4.16	27	10
C4	7.1	0.2	-54	96	0.9	0.2	1.89	0.7	0.00	0.00	0.84	0.25	7.91	2.85	107	23
D1	6.9	0.2	13	105	0.6	0.2	0.31	0.1	1.18	1.49	0.09	0.16	1.37	0.48	19	3
D2	7.2	0.2	-4	100	0.7	0.2	0.17	0.1	0.01	0.04	0.08	0.03	0.85	0.28	16	4
D3	7.4	0.4	-57	82	0.7	0.2	0.94	0.3	0.00	0.00	0.37	0.07	0.96	0.65	16	3
D4	7.4	0.4	-15	127	0.8	0.3	1.39	0.4	0.00	0.00	0.58	0.16	1.92	0.92	34	4
P1	7.2	0.2	-58	84	0.8	0.2	1.79	0.7	0.01	0.03	0.38	0.13	4.83	1.08	434	80
P2	6.4	0.4	90	78	0.6	0.3	0.26	0.3	9.53	5.48	0.04	0.03	0.91	0.86	66	28
W1	7.3	0.5	95	68	0.0	0.0	0.73	0.3	9.44	2.33	0.09	0.05	0.56	0.46	90	10

Appendix 2. Spatial and temporal mean values (with standard deviation) of various inorganic chemicals measured at R-21.

Sampling date	pH		Eh (mV)		WT (m)		NH_4-N (mg/L)		PO_4-P (mg/L)		Fe (mg/L)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
16/11/93	5.2	0.3	331	106	3.4	0.2	0.56	1.1	0.0	0.0	0.29	0.9
02/10/94	5.1	0.3	170	43	3.2	0.2	0.37	0.4	0.1	0.1	5.25	2.7
19/01/94	4.8	0.2	158	66	3.1	0.3	0.49	0.3	0.1	0.1	5.67	3.0
03/02/94	5.0	0.2	173	64	3.0	0.2	0.62	0.5	0.1	0.0	4.96	3.0
14/02/94	4.9	0.2	238	25	2.6	0.2	0.75	0.9	0.0	0.0	5.87	3.3
28/02/94	5.1	0.2	267	61	2.2	0.2	0.59	0.6	0.0	0.0	7.72	3.5
20/03/94	5.1	0.3	105	57	3.3	0.2	0.41	0.3	0.0	0.0	3.33	2.1
28/03/94	5.2	0.3	331	106	3.4	0.2	0.35	0.3	0.0	0.1	4.28	3.1
11/05/94	5.1	0.2	284	25	3.2	0.2	0.34	0.4	0.1	0.1	5.58	3.3
18/05/94	4.9	0.2	286	44	3.3	0.2	0.33	0.3	0.0	0.1	2.77	2.8
24/05/94	5.1	0.2	300	45	3.2	0.2	0.35	0.3	0.1	0.1	4.90	3.0
31/05/94	4.7	0.5	231	108	3.2	0.2	0.35	0.4	0.0	0.1	4.02	2.5
07/06/94	4.8	0.5	165	266	3.2	0.2	0.43	0.4	0.0	0.0	3.89	2.5
14/06/94			19	100	3.3	0.2	0.52	0.5	0.1	0.1	3.38	2.3

Sampling point	pH		Eh (mV)		WT (m)		NH_4-N (mg/L)		PO_4-P (mg/L)		Fe (mg/L)	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
A1	4.9	0.3	220	109	3.1	0.4	0.25	0.1	0.037	0.0	2.89	1.2
A2	4.9	0.3	212	108	3.0	0.4	0.26	0.1	0.051	0.1	3.03	1.4
A3	5.0	0.2	196	103	3.0	0.4	0.38	0.1	0.040	0.0	5.68	2.3
A4	4.9	0.3	186	118	2.9	0.4	0.34	0.1	0.042	0.0	3.94	2.3
B1	5.0	0.4	175	116	3.3	0.4	0.51	0.2	0.057	0.1	7.18	2.7
B2	4.9	0.3	211	167	3.3	0.3	0.47	0.3	0.034	0.0	6.00	5.5
B3	5.4	0.2	232	130	2.8	0.4	1.19	0.8	0.134	0.1	6.14	4.3
B4	5.1	0.4	211	100	3.2	0.4	0.60	0.5	0.060	0.1	5.49	3.5
C1	5.1	0.3	217	137	3.4	0.3	0.32	0.1	0.050	0.1	5.13	2.6
C2	5.1	0.5	197	93	3.3	0.3	0.36	0.2	0.021	0.0	2.90	2.8
C3	5.2	0.2	269	173	3.1	0.3	0.48	0.4	0.048	0.0	8.29	2.9
C4	4.7	0.2	228	98	3.0	0.3	0.18	0.1	0.027	0.0	2.28	1.0
D1	4.8	0.2	211	93	3.3	0.3	0.18	0.1	0.016	0.0	2.17	1.2
D2	4.8	0.2	207	192	3.2	0.3	0.19	0.1	0.044	0.1	3.28	2.6
D3	5.1	0.3	265	131	3.1	0.3	0.44	0.3	0.063	0.1	6.01	2.2
D4	4.8	0.1	212	97	3.0	0.2	0.24	0.1	0.037	0.1	3.64	1.7
DW	5.1	0.2	259	217	—	—	0.15	0.1	0.036	0.1	1.81	1.0
ST	5.1	0.2	224	118	2.8	0.4	1.76	1.0	0.175	0.2	3.73	3.2

Evaluation of Field Sampling Methods for Soil Water Quality

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Abstract

A field study was conducted on a horticultural farm on the northern Swan Coastal Plain, Western Australia. The objective was to compare and evaluate chemical composition (in terms of NO_3, P, Cl) of soil water collected by the three most acceptable methods: i) ceramic suction cup sampling, ii) drainage lysimetry and iii) gravimetric soil sampling.

The ceramic cups were installed either by surrounding the cup with silica flour slurry, or sand. The experiment was arranged as a nested factorial design where the three factors were: i) sampling method, ii) location, and iii) time of sampling.

The study shows that the method of installation (use of silica slurry vs sand) of ceramic cups had a negligible effect on the mean values, but it appears that by using silica the uncertainty of the mean was reduced considerably. The average concentrations of NO_3-N and Cl were higher for soil sampling than those measured by ceramic cups. The mean values of all the three constituents (NO_3, P, Cl) were higher for ceramic cups than for lysimeters. Although seasonal consistency was maintained for the majority of the period, the differences were statistically significant for NO_3 only. The order of methods with respect to extent of variability associated with mean measured values was: soil sampling>> ceramic suction cup with sand > ceramic suction cup with silica slurry>> lysimeters.

RELIABLE techniques for collecting data on soil water quality are crucial for studying the transport of pollutants through soils to aquifers and streams. Different types of sampling equipment are used in determining soil water quality of the unsaturated zone in the field. The most widely used device is the porous ceramic cup. Other devices include the free draining pan lysimeter, zero tension sampler, fritted glass cup, wick lysimeter, hollow tube bundles, and direct (gravimetric) soil sampling. A good understanding of the limitations and reliability of the equipment is necessary to interpret data satisfactorily.

The three most acceptable sampling methods are ceramic suction cups, drainage lysimeters, and gravimetric soil sampling. The objectives of this study were:

- to compare and evaluate chemical composition (in terms of NO_3, P and Cl) of soil water collected under field conditions by the above three methods, and
- to provide some practical guidance in soil water sampling.

The Basis of Soil Water Sampling

Soil water through a filter such as a ceramic cup is collected by applying negative pressure within the cup. Ceramic cups thus collect gravity and capillary water, while the gravity lysimeters collect only the gravity water (<0.01 MPa). When a vacuum is applied in a ceramic cup, it is easier to collect water in larger pores than in smaller pores. The concentration of chemicals may vary greatly with distance from soil particles. As water composition is also a function of pore size, the composition of the soil solution is not homogeneous (England 1974). The recommended upper limit of suction to be applied for collecting samples is 0.1 bar (0.01 MPa), particularly if the intention is to collect free-draining water (Severson and Grigal 1976).

Gravimetric soil sampling involves the addition of a known amount of distilled water to saturate the sample. The samples are thoroughly mixed and soil water is extracted for chemical analysis. The chemical concentrations are converted to original field water content basis. It is assumed that the solutes are fully dissolved. This method measures the average chemical concentration of the soil water that is left in the soil matrix after certain

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drainage, but is limited by expressing soil water composition in smaller pores than either the ceramic cup, or the lysimeter.

Location and Method

The experiments were conducted on a horticultural farm on the northern Swan Coastal Plain, Western Australia. The soil association on this location is classified as Spearwood sand (McArther and Batteney 1960) which is derived from limestone and is aeolian in origin. It contains >95% particles in the sand fraction.

During experimentation, pumpkins were grown on the site. Overhead sprinklers were used to irrigate the crop from the aquifer beneath. The three types of sampling equipment used were gravity lysimeters, ceramic cups and soil sampling equipment. The ceramic cups were installed at five locations around the three large gravity lysimeters. The design and installation procedure of lysimeters has been described in Sharma et al. (1991), and Shimojima and Sharma (1991).

The ceramic cup assembly consisted of a semi-spherical cup (Soil Moisture Equipment Co., Santa Barbara, California, U.S.A.) which was 50 mm in diameter and attached to PVC plastic tubes of varying length. The ceramic cup was 60 mm long with a surface area of 0.0094 m^2. The cups were installed in two ways, either they were surrounded by silica slurry, or they were backfilled with dry sand. All the samplers were installed close to the lysimeters. Soil cores 52 mm in diameter were taken down to a depth about 5 cm deeper than the desired installation depth. Ten ceramic cups were installed at depths of 50 cm, 100 cm and 250 cm. Five cups had silica slurry around them, the rest had dry sand as backfill. Continuous tamping was maintained to ensure good contact and sealing of the cored hole. The samplers were allowed to settle for about two weeks before suction (0.06 MPa) was applied for soil water sampling.

The intake rate of the ceramic cups was measured before installation and after completion of the experiment. Water samples from both ceramic cup and lysimeter samples were collected every two weeks. At selected times soil core samples were collected from specified depth intervals. Soil water was extracted and analysed.

The four sampling methods compared are:

- Method 1: Ceramic cups with sand around the porous cups
- Method 2: Ceramic cups with silica slurry around the porous cups
- Method 3: Gravimetric soil sampling
- Method 4: Lysimeters (at 250 cm depth only)

Analysis of variance (ANOVA) technique was used to compare the results of this study. The experiment was arranged as a nested factorial design where the three factors were: i) sampling method, ii) location and iii) time of sampling. In the ANOVA the main effects were sampling method and location; the nested factor was time of sampling.

Results and Discussion

While making comparisons among various methods, it is important to recognise the extent of temporal and spatial integration of the equipment for each sampling method. In this regard the following points should be considered.

Lysimeter sampling is continuous in time, while ceramic cup sampling most likely does not integrate evenly over the entire sampling period. Soil samples only provide information at the time of collection.

Lysimeter samples represent flux concentration, whereas ceramic cup and soil samples represent resident concentration. Flux concentration can be visualised as the mass of solute crossing a plane, perpendicular to the solute flux direction, divided by the volume of water that crosses the plane over the same period. The resident concentration is defined as the mass of solute per unit volume of soil.

There is a huge difference in the size of the equipment. Lysimeters sampled from a cross sectional area of 1.77 m^2, whereas the surface area of the ceramic cup was 0.0094 m^2. The soil samples were 50 g in weight (~25 cm^3).

The lysimeters were free-draining. They sampled water held at suction below field capacity (<0.01 MPa), while ceramic cups extracted soil water held at suction up to 0.06 MPa. Soil water extracts, on the other hand, represent mixed water from the soil matrix, and mostly exclude free draining water.

For comparisons to be made, the validity of the results from the three techniques need to be evaluated in light of factors encountered in the installation and operation of techniques. Examples of these are variations in sample size, extent of temporal integration, spatial variability, and disturbance caused in the installation of equipment. Because of such complexities, it is doubtful whether rigorous statistical analyses could be valid when applied to field data to differentiate the technical effects alone. However, based on the estimated confidence limits around measured mean values, and examination of consistency in the temporal trends, it is believed that some useful generalisations can be drawn. Mean (arranged over space and time) and standard deviations of each of the chemical parameters measured by the four methods are summarised in Table 1. Ratios of these parameter means (based on a pair of selected methods) are also presented in Table 1. Also indicated are the cases when the differences between the various methods are significantly different based on ANOVA.

Effects of the Installation of the Ceramic Cup

Time variations of mean values and their uncertainty (±1 x s.d.) of NO_3-N, Cl and P measured by Method 1 (M1, sand) and Method 2 (M2, silica slurry) are presented in Figure 1. Table 1 and Figure 1 show that there is considerable uncertainty associated with the estimated mean at a given time. Overall, the uncertainty is much larger with M1 than M2. P values were consistently higher for M2, while NO_3-N and Cl values for M1 were consistently higher except for the first two to three observations.

The mean sample volumes collected and mean residual vacuum left in various ceramic samplers used in the two methods over the season were not significantly different for the two methods. Overall, the change in intake rate was found to be negligible, suggesting that physical clogging of the cups did not occur. On average the intake rate of cups installed in silica (M2) increased by about 6%, while those installed with sand (M1) decreased by 5%, however, such small changes are expected to be within experimental error.

These observations suggest that the method of installation had a small effect on the overall mean values of measured chemical parameters. However, the uncertainty of the estimated mean was considerably reduced when silica slurry (M2) was used (Fig. 1).

Comparison of Ceramic Cup and Soil Sampling

A comparison of ratios of mean NO_3-N and Cl values at three depths (50 cm, 100 cm and 250 cm) obtained through M1 and soil sampling (M3) is presented in Table 1. Time variations of mean values (±1 x s.d.) are compared in Figure 2.

In almost all cases mean values of both NO_3-N and Cl concentrations were higher for the M3 than for the ceramic cup method. The differences between methods for NO_3-N and Cl were found to be statistically significant at 100 cm depth only. In most cases, seasonal consistency is maintained through time. Uncertainties associated with means for M3 were much higher than for M1, because locations for five core samples at each sampling time were different in the case of M3, while they were the same for M1.

Table 1. Overall mean (mg/L) standard deviation (in parentheses) and ratios of mean concentrations of samples collected by different methods.

Depth (cm)	Variable	Method 1	Method 2	Method 3	Method 4	M1/M2	M1/M3	M2/M3	M1/M4	M3/M4
50	NO_3-N	88.5	83	101.4		1.07	0.87	0.82		
		(49.4)	(30.4)	(33.7)						
	Cl	123.2	91.8	175.9		1.34	0.70	0.52		
		(81.5)	(51.9)	(199.7)						
	P	6.08	7.38			0.82				
		(2.35)	(2.0)							
100	NO_3-N	76.5		141.6			0.54*			
		(30.8)		(96.4)						
	Cl	91.7		472.6			0.19*			
		(151.5)		(409.5)						
250	NO_3-N	110.6		156.9	87.3		0.70		1.27*	1.8
	Cl	173.5		175.9	159.7		0.99		1.09	1.1
		(189.8)		(199.7)	(149.0)					
	PO_4-P	0.02			0.007				2.86	
		(0.021)			(0.011)					

M1 = Method 1 = Ceramic cups with sand slurry around the porous cup
M2 = Method 2 = Ceramic cups with silica slurry around the porous cup
M3 = Method 3 = Gravimetric soil samples
M4 = Method 4 = Lysimeter samples

* indicates that statistically significant difference exists at $\alpha = 0.05$

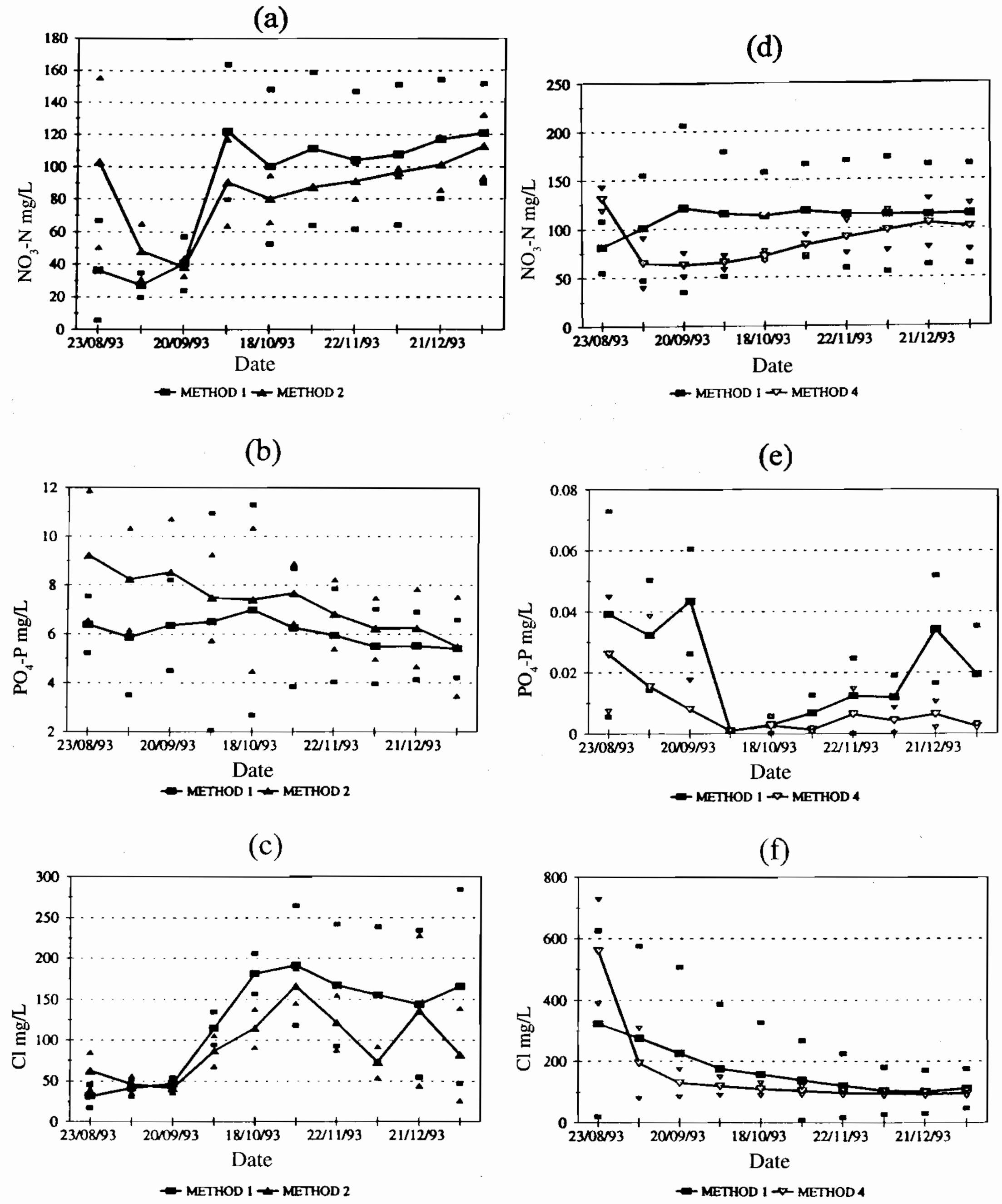

Figure 1. Comparison of temporal variation (shown with 1 x S.D.) in the mean concentrations of NO_3-N, PO_4-P and Cl measured by Method 1 (sand) and Method 2 (silica) (a,b,c,) and measured by Method 1 (sand) and Method 4 (lysimetry) (d, e, f).

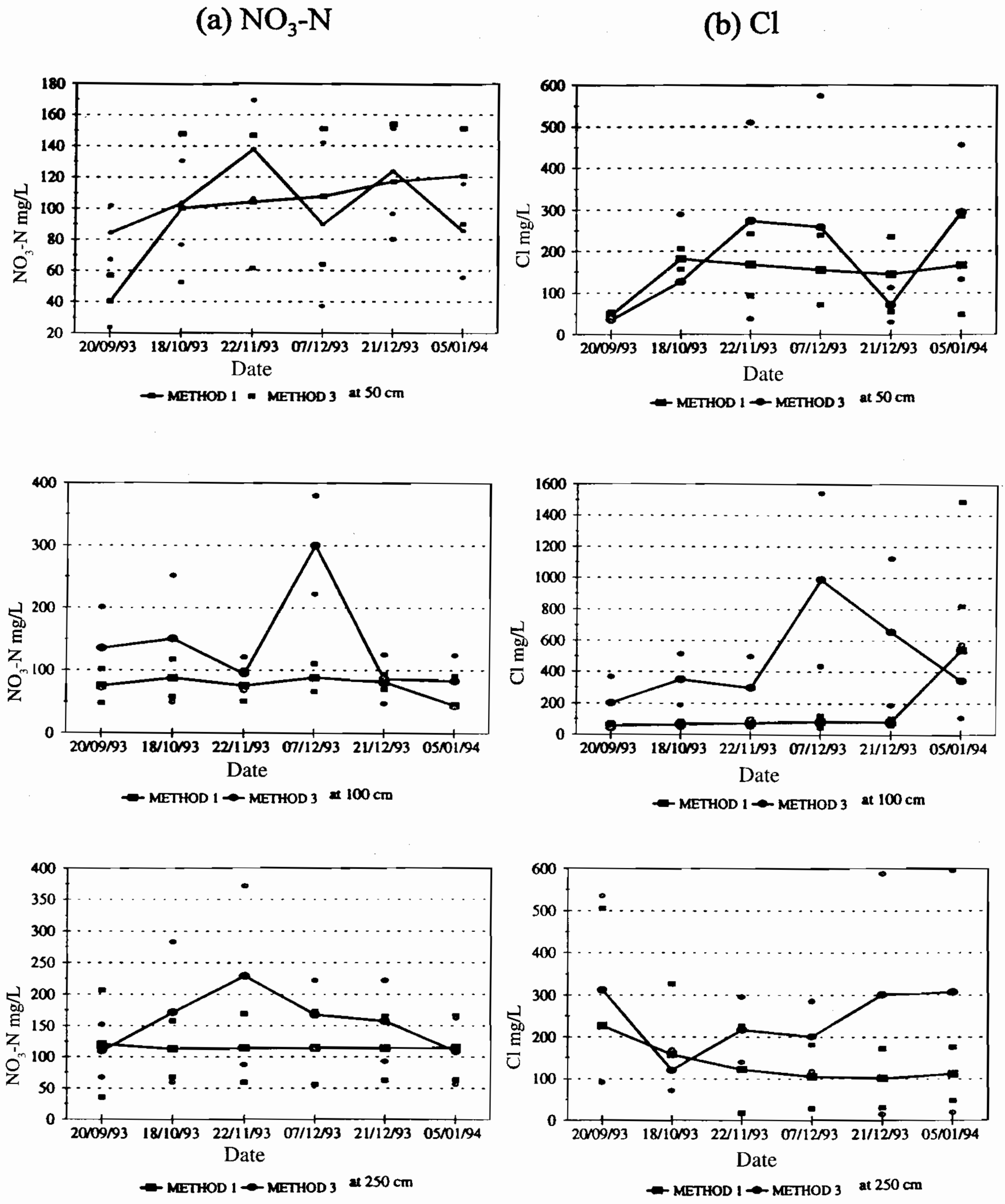

Figure 2. Comparison of temporal variation in the mean concentrations (shown with 1 S.D.) of (a) NO$_3$-N and (b) Cl and measured by Method 1 (sand) and Method 3 (soil sampling).

If ceramic cups were sampling both matrix and macropore water, these results appear logical. Water extracted by the soil sampling method would not be sampling the macropore water, which would have a lower concentration.

Differences in ceramic cup versus soil samplings have been reported previously (e.g. Alberts et al. 1977; Barbarick et al. 1979; Webster et al. 1993). Because anions can be excluded from a portion of the soil solution (closer to soil particles) due to electrical repulsion (de Haan and Bolt 1963) it is expected that anion concentrations of soil solution held at lower tension is usually higher, and cation concentration lower than the average of the entire soil solution (Smith and Davis 1974). Barbarick et al. (1979) found that Na and Ca concentrations based on soil sampling were high in the ceramic cup solutions. Alberts et al. (1977) also found that in a 3 m profile of a silt loam soil, NO_3-N concentrations based on soil coring were higher in two out of three years if using ceramic cups, but the differences were significant only for the first year. On the other hand, Webster et al. (1993) reported that NO_3-N concentrations were significantly lower with soil coring than with ceramic cups. Furthermore, the results from soil coring had much larger uncertainty and were not consistent in time.

Comparison of Ceramic Cup and Lysimetry

Figure 1 (d,e,f) compares the time variations of areal means (±1 x s.d.) of NO_3-N, Cl and P measured by the two methods. It is obvious that uncertainty associated with mean values for ceramic cup (M1) is several times larger than for lysimeter (M4). It is interesting that seasonal consistency is maintained. For the majority of the observation period, mean values of all three anions are higher for the ceramic cups than the lysimeter. However, because of doubt about the ceramic cup means, the differences were found to be statistically significant for NO_3-N only.

Based on the anion exclusion principle discussed earlier, it is expected that the lysimeter samples should have higher concentrations of anions (NO_3 and Cl) than the ceramic cup samples, as the former is supposed to sample water away from the soil surface. However, such expectations are not always supported by field observations because of many complicating factors in the field. For example, Haines et al. (1982) found that not only were average NO_3-N and Cl concentrations higher in zero-tension than in tension lysimeters, but also average concentrations of cations (NH_4, K, Ca) were higher. Instantaneous soil solution nutrient concentrations are likely to be an interacting function of microbial activity, plant uptake, soil water content and various factors affecting field spatial variability.

In the northern Swan Coastal Plain, the overall mean value of NO_3-N for the season was about 27% larger for the ceramic cup than the lysimeter. These differences could be explained by the differences in the volume of soil samples. Ceramic cups were placed directly beneath the rows near the plants where NPK fertiliser was applied, and irrigation water was applied evenly by sprinklers. On the other hand, with a surface sampling cross sectional area of 1.77 m^2, each lysimeter collected water beneath rows as well as furrows. Assuming a mainly one-dimensional flow, it is expected that NO_3-N, Cl, P originating from fertiliser will show higher concentrations beneath the rows (sampled by ceramic cups) than beneath the furrows. Furthermore, it is expected that ceramic cups will collect proportionately smaller amounts of macropore water than lysimeters.

Recently Webster et al. (1993) compared free-draining lysimetry, ceramic cups and soil core methods for studying NO_3-N concentrations in two light-textured soils under cropping. Overall, NO_3-N concentrations from soil core samples were lower than the other two methods, and the differences were significant with respect to free-draining lysimeters. The results from lysimeters and ceramic cups were reasonably compatible, while those from soil samples differed considerably, but with no consistent bias.

Conclusions

The suitability of a sampling technique depends on factors such as soil type, soil moisture condition and availability of resources. Comparison of various methods should consider the primary objective of sampling. The objective in this case was to measure flux-averaged concentrations of ions (primarily NO_3-N, Cl, P, NH_4-N) in the water draining below a specified depth in the field. Provided adequate precautions are taken in the design and installation of lysimeters, lysimetry is probably the best available technique (Sharma et al. 1991). However, lysimeters are expensive, and under certain conditions it may not be feasible to use them. Therefore, alternative methods are used, for example, ceramic cups and soil sampling. When the objective is to measure flux-averaged concentration of a chemical in the leaching water, the soil sampling technique measures concentration of water which has been left in the soil. Ceramic cup sampling approximates the leachate concentration. In addition to the dissimilarities in the temporal and spatial integration of the sample in the soil profile, there are a large number of factors which can affect ceramic cup results. Some of these factors include possible sorption by cup material (teflon cups are considered the best, but very expensive) size of the cup (Silkworth and Grigal 1981) amount and duration of suction applied (Severson and Grigal 1976; Hansen and Harris 1975) plugging (Hansen and Harris 1975; Silkworth and Grigal 1976) and clogging (Everett 1990; de Hann and Bold 1963; Smith and Davis 1974; Shimojima and Sharma 1991).

Because of the variation in sample size and a large number of factors affecting the results, rigorous analysis of comparison of field methods is unlikely. Laboratory studies can evaluate the effect of one or more factors under controlled conditions, but they cannot provide conclusive guidance for field use. Therefore, comparisons made based on field studies have to be used, but they should be considered as guidance only.

Conclusions based on our studies are summarised below:

- The method of installation (use of silica slurry vs sand) of ceramic cups had a negligible effect on the mean values, but it appears that by using silica the uncertainty of the mean was reduced considerably. It is possible that the use of silica rather than sand around cups may reduce plugging/clogging.
- In addition to a general temporal consistency being maintained, the average concentrations of NO_3-N and Cl were higher by soil sampling than by the ceramic cup method. These differences were significant only at one out of three measuring depths. This was primarily due to significant uncertainty associated with mean values, especially with the soil sampling method.
- The mean values of all the three anions (NO_3-N, Cl, P) were higher for ceramic cups than for lysimeters. Although seasonal consistency was maintained for the majority of the period, the differences were statistically significant for NO_3-N only, mainly because of uncertainty associated with ceramic cup values.
- Based on our observations the order of methods regarding the extent of uncertainty associated with the mean values was: soil sampling (M3) >> ceramic cups with sand (M1) > ceramic cups with silica slurry (M2) >> lysimeters (M4). Thus the soil sampling method is the least reliable. Provided that certain precautions are taken in the selection and installation of ceramic cups, they can provide useful information for solute transport in the field. The use of silica slurry around the cups is likely to improve the sampling.

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Impact of the Use of Agrochemicals on Water Resources in Southern Thailand

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Abstract

Thailand has experienced a rapid increase in the use of agrochemicals during the past 20 years. Low levels of pesticides have been detected in vegetable crops, but there is little information available about contamination of rural water resources used for drinking. This paper highlights the need to determine present levels of agrochemical contamination in water in rural Thailand and to find ways to reduce agrochemical buildup in the environment.

THE USE of agrochemicals has become an integral part of agricultural production in Thailand. It is estimated that 15–30% of annual crop production is lost to pests, diseases, and weed competition, consequently, the demand for agrochemicals has increased rapidly over the past 20 years. Chemical fertilisers used in Thailand during 1986–90 are shown in Fig. 1a. The quantity of pesticides imported during 1986–90 is shown in Fig. 1b. Heavy application of these pesticides have caused many problems which have become increasingly serious.

Pesticide Poisoning of Farmers

Pesticides can be introduced into the human body by direct exposure, or indirect exposure by consuming food containing their residues. In 1985, a total of 5458 pesticide poisoning cases were recorded, including 384 deaths (Table 1). Most of the poisoning cases were farmers who had been exposed to pesticides for a long period of time.

Pesticide Residues in Agricultural Products

Studies of agricultural products show a relatively large number of samples were contaminated with residues of organochlorine pesticides (Department of Agriculture 1980; 1983). Furthermore, the results of an extensive survey reported in Tayaputch (1988) revealed that a relatively large number of agricultural products such as vegetables, fruits, grains, oil crops, field crops, meat and eggs were contaminated with residues of organochlorines (BHC, heptachlor and its epoxide, aldrin, dieldrin, endrin, DDT and metabolites), organophosphates (dimethoate, methyl parathion, parathion, methamidophos, mevinphos, monocrotophos, diazinon and dichlorvos) and carbamates (methomyl and carbofuran).

Hazards in the Environment

The main environmental problem caused by agrochemicals is pesticide pollution in bodies of water. Surveys and monitoring of pesticide residues in the environment have been conducted by many researchers since 1976 (Tayaputch 1988). The results obtained by the Division of Agricultural Toxic Substances showed that trace amounts of organochlorines were found in 50% of 1644 water samples, and 90% of 602 sediment samples collected from 30 main canals, rivers, and water reservoirs used for drinking water. Although these chemicals were banned from use in Thailand in 1981,

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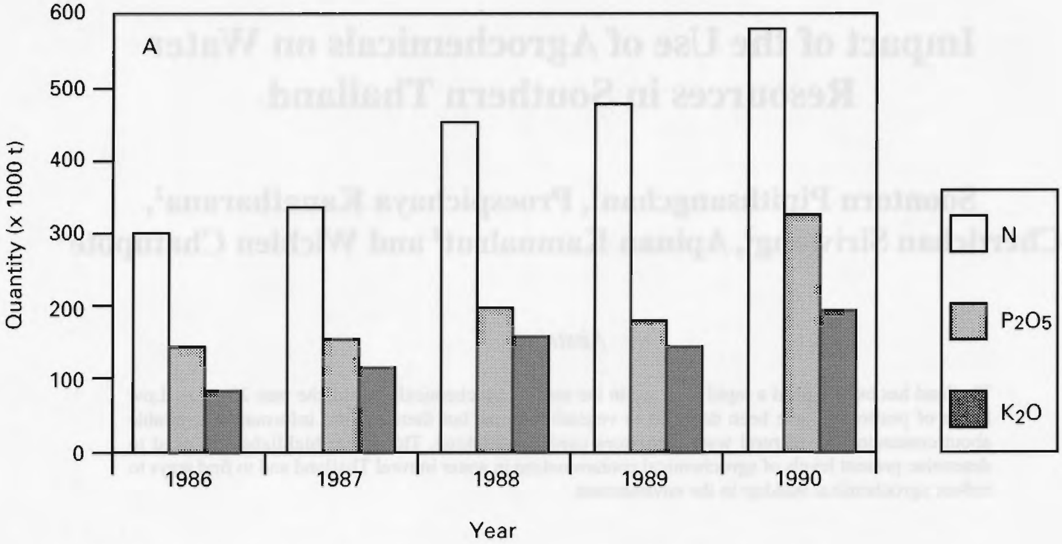

Figure 1A. Chemical fertilisers used in the form of major plant nutrients in Thailand, 1986–90.

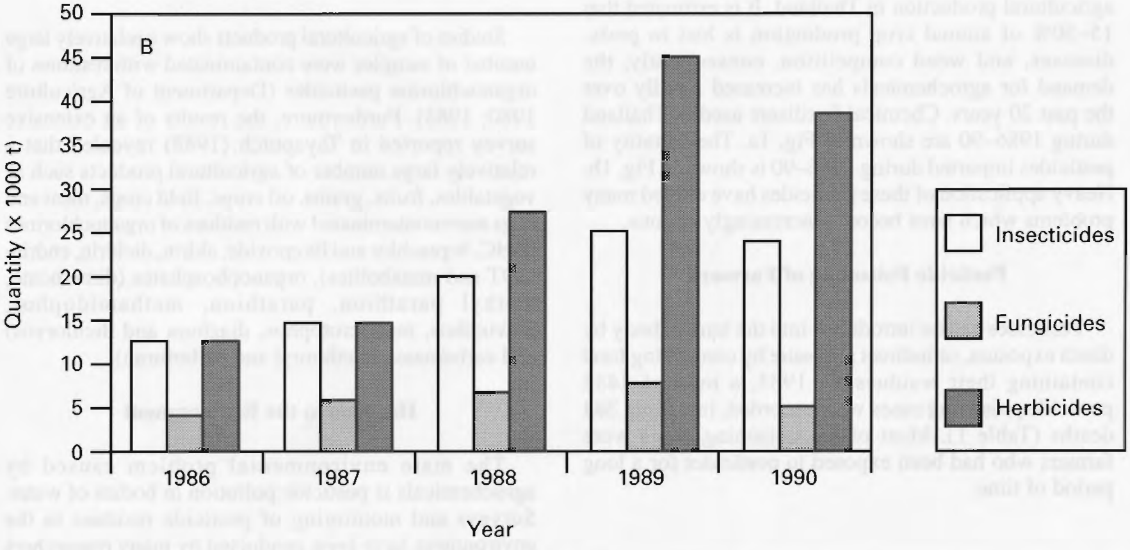

Figure 1B. Quantity of pesticides imported into Thailand, 1986–90.

Source: Office of Agricultural Economics, 1992.

Table 1. Number of poisoning cases and deaths caused by pesticides in each region of Thailand in 1985.

Region (Provinces)	Poisoning cases		Deaths		% Deaths per Poisoning
	No.	%	No.	%	
1. Central (22)	2570	47	163	43	6.3
2. Northern (12)	1822	33	143	37	7.8
3.Northeastern (14)	667	12	50	13	7.5
4. Southern (12)	399	8	28	7	7.0
Total	5458	100	384	100	

Source: Office of National Environmental Board 1985.

these persistent organochlorine pesticides appeared to have been widely distributed in the environment and living organisms. It was found that 90% of fish and shellfish from market places and water sources in central Thailand, and green mussels from the Gulf of Thailand, contained residues of DDT, aldrin, and dieldrin (Tayaputch 1988; Siriwong et al. 1991).

A Sample Case in Southern Thailand

The area of Rattaphum watershed was selected as a regional case example. It is on the eastern side of southern Thailand, about 35 km north of Hat Yai.

Physiographical Features

The western border of Rattaphum extends over parts of the Khao Bantad mountain range, which is covered with forest. On the eastern side there is low-lying terrain connecting the Outer Songkla Lake used by farmers for paddy fields. (Fig. 2). Humic gley and podzols are the main soil types of the lowland and foothills, respectively.

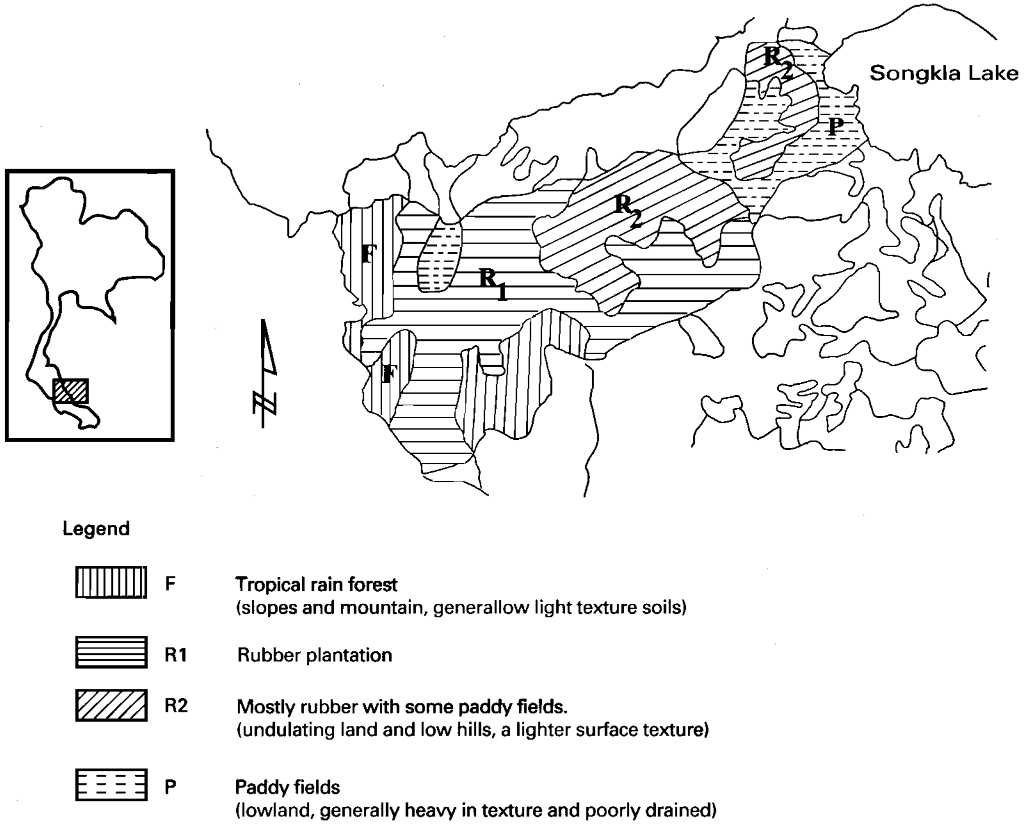

Figure 2. Land use map of Rattaphum Watershed.

Its average annual rainfall is about 2000 mm, with over 75% falling between October and January. February and March are very dry.

General Agricultural Land Use Patterns

There are four major types of agricultural land use patterns: rubber plantations in the higher upper terrain (approx. 50%), paddy fields (approx. 20%), a mixture of paddy fields and vegetables, and some field crops in low-lying areas (approx. 15%).

The Use of Agrochemicals

Agrochemicals are widely used in many types of agricultural production, but it is not possible to establish the total amount used. However, it may be useful to look at the amount distributed by the government for pest control as this will reflect the agrochemicals used. The amount of pesticide and rodenticides distributed by the Ministry of Agriculture and Co-operatives to farmers in the southern provinces is shown in Table 2.

In a report by the Division of Agricultural Toxic Substances, Department of Agriculture (Imphitask 1994) heavy use of agrochemicals, especially pesticides, was found in the vegetable growing areas listed in Table 2. Pesticides available for sale in the area are paraquat, glyphosate, alachlor, monochrotophos, methomyl, chlorpyrifos, MTMC, isoprocarb, carbosulfan, carbaryl, BPMC, methamidophos, carbendazin, mancozeb, cyhalothrin, sumicidin and the microbial insecticide *Bacillus thuringiensis*. Analysis of pesticide residues in several kinds of vegetable products was also done by the Department of Agriculture, resulting in all herbicide residues being lower than the codex Maximum Residue Limit (MRL) of the Food and Agriculture Organisation (FAO) and the World Health Organisation WHO (Table 3). It is anticipated, however, that bio-accumulation of residues would occur through continuous application of the pesticides.

Application of fertilisers was common in rubber, rice and vegetable production. Vegetable growing required higher amounts of fertilisers than other crops. In the area where farmers grew vegetables only, an average of three crops per year was commonly seen, hence the need for fertilisers has tripled. Most farmers in the area used compound fertilisers of the grade 15-15-15 NPK with an amount of 250 kg/ha per crop. Other fertiliser grades used were 13-13-21 NPK, 16-8-4 NPK, and urea. The amount of fertiliser applied depended upon the kinds of vegetable grown. For example, an average of 400 kg/ha per crop was used in growing cauliflower (Tungwa et al. 1989). Other vegetables that were also commonly grown with intensive management were cucumbers, yard long beans, chillies, gourds, and eggplants.

Table 2. Quantity (tonnes) of pesticides and rodenticides distributed by the Government in the four southern provinces during 1991-1993.

Pesticides	Year	Songkla	Province NarathiwasPattani	Yala	Total	
	1991	1202	600	400	273	2475
Insecticide	1992	4765	1362	990	696	7813
	1993	1897	420	440	132	2889
	1991	1670	20	222	182	2094
Fungicide	1992	2236	34	214	228	2712
	1993	2525	10	216	245	2996
	1991	-	-	300	-	33
Herbicide	1992	900	-	600	-	1500
	1993	-	-	900	-	900
	1991	3203	1874	1290	664	7031
Rodenticide	1992	2501	1252	520	470	4743
	1993	4811	1208	1440	676	8135

Table 3. Pesticide residues found in vegetables harvested in August 1994 in Rattaphum District, Songkla Province as compared to the codex MRL.

Vegetable	Pesticide name and amount (ng/mL)		FAO/WHO codex MRL (mg/L)
Leafy onion	Diazinon	0.3	0.7
Leafy pai-sai	Total DDT	1.5	1.0
Chinese kale	Total DDT	0.001	1.0
Lettuce	Total DDT	0.01	1.0
Morning glory	Total DDT	0.61	1.0
Chinese cabbage	Total DDT	1.0	1.0
Yard long bean	Total DDT	1.85	1.0

The Use of Water Resources

The main sources for drinking water and daily consumption are from shallow and deep wells, or water is pumped from deeper groundwater sources. Vegetables are irrigated using water from streams, canals, and rivers. However, in areas near the lake, such as Ban Bang Riang where the water becomes brackish during the dry season, underground water is used. About 30% of the villagers used water from shallow wells for both drinking and irrigating vegetables (Tungwa et al. 1989) while 40% had separate shallow wells for each purpose. The rest used water from canals, reservoirs, and rivers to irrigate the vegetables. However, they reported an increasing tendency to use groundwater for both purposes. Average depth of groundwater was between 30 and 70 m. Vegetable growing areas were usually located near the water sources. About 60% of the villagers had their vegetable fields near their houses, 30% had vegetable fields far from from their houses, and the rest grew vegetables in their rice fields after harvesting rice.

Impact of Agrochemicals on Water Resources

There is little information about the contamination of agrochemicals in water resources in southern Thailand. Analysis of water quality from different water resources throughout southern Thailand reported by Na Chiangmai (1988) stated that the water in Songkla Lake contained the pesticide BHC at a level exceeding the standard of water quality for surface water (0.02 ng/mL). Others contaminants, like DDE, DDD, heptachlor, epoxide, aldrin, endrin, NO_3-N, P, and other substances were found in trace amounts (Table 4).

Table 4. Chemical residues found in water resources in Southern Thailand (Na Chiangmai 1988).

Chemical residues	Concentration
Phosphate	0.003-0.23 (mg/L)
Copper	0.0-0.08 (mg/L)
Lead	0.0-0.82 (mg/L)
Zinc	0.0-0.055 (mg/L)
BHC	0.0-10.62 (ng/mL)
DDD	0.0-0.92 (ng/mL)
Aldrin	0.0-0.016 (ng/mL)
Endrin	0.0-0.002 (ng/mL)

Bunvanno (1992) found that pesticide residues of DDT and its metabolised products in outer Songkla Lake during September 1991 to February 1992 were high in concentration, and similar results were obtained by Kanatharana (1993) who monitored water quality in the lake up to November 1992.

Recently, Kanatharana and Eiatjuice (1994) found that DDT and its metabolised products in Klong Wad Canal, connecting the Inner Songkla Lake to the Outer Songkla Lake, were in the range of 0 to 49 ng/mL. So far, there have been no studies to assess the quality of groundwater affected by agrochemicals in this area.

Conclusion

There is a clear evidence that agrochemicals have become an increasingly serious problem in Thailand. Pesticides are found to be the most hazardous, especially as a result of intensive cropping such as vegetable growing. We have little information concerning the agrochemical contamination of the water resources used for drinking purposes in rural areas. Thus, urgent investigation into the quality of water affected by the use of agrochemicals is needed to establish the present level of contamination. In addition, safety measures for using agrochemicals, and means of preventing contamination must be devised to improve the living conditions of the rural people and the environment.

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Nitrate in Javanese Well Water: Occurrence and Origins

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Abstract

Surveys of household wells in East Java showed that millions of people on the island drink water with a NO_3-N level greater than the World Health Organisation standard of 10 mg/L. Seventeen per cent of the wells surveyed in the dry season in 1991, and 28% of wells surveyed in the wet season in 1992 had NO_3-N levels greater than 10 mg/L. The distribution of nitrate levels in groundwater within a settlement indicated that N fertiliser applied to rice fields was not likely to be the main source for the nitrate found in the groundwater underneath villages. It is argued that human waste and to some extent animal manure, within a settlement is the main contributor of N to the point sources for nitrate plumes in the groundwater underneath settlements.

INDONESIAN lowland rice farmers use large quantities of nitrogenous fertilisers, mainly in the form of urea, to obtain high grain yields. In view of agricultural pollution problems in western countries, it was suspected that such high N inputs could lead to nitrate pollution of groundwater. This in turn, could create a health hazard as the majority of the rural population in Indonesia obtains drinking water from wells that draw from shallow groundwater sources.

The main potential health risks from nitrate ingestion are thought to be methaemoglobinaemia (Super et al. 1981; Shearer et al. 1972) oesophageal and stomach cancer (Forman 1989; Siddiqi et al. 1992) insulin-dependent diabetes mellitis (Kostraba et al. 1992) and thyroid hypertrophy (Maanen et al. 1994). The incidence of goitre and insulin-dependent diabetes is high in Java and increasing.

In view of the health risks, the Indonesian Center for Policy and Implementation Studies (CPIS) funded a project to carry out two surveys in east Java to assess the nitrate concentrations in groundwater mainly drawn from household wells, but it also included rivers, sawahs, and irrigations canals. Based on the results of the first survey (dry season 1991) sources of the nitrate found in many wells were postulated and investigated in a second survey (wet season 1992). A more detailed study was undertaken in Kota Gede near Yogyakarta, central Java, in cooperation with the University of Gadjah Mada in January/February and September 1994. The main results from the survey are reported in this paper.

Methodology

The survey in East Java involved random selection of villages and towns, and random selection of household wells within these settlements. In the wells, the depth of the water table was measured and a water sample collected. The nitrate concentration in each sample was determined immediately after collection, using the Merck Nitrachek method. When the concentration was greater than 10 mg/L NO_3-N, a second sample was collected and the nitrate concentration determined the same way under temperature controlled conditions, using a series of standards for calibration.

In some villages, soil cores to 2 m depth were collected in 25 cm increments and the nitrate concentration in each increment was determined on the same day. Total N values in topsoils were determined

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using a Kjeldahl method that includes nitrate (Bremner and Mulvaney 1982). Representative samples of topsoils were incubated anaerobically for one week at 37°C, using the method described by Keeney (1982) to measure the mineralisation rate of the soil organic N.

In Kota Gede, Central Java, detailed studies were made in the administrative unit RW5, containing approximately 125 household wells. In addition to determining the nitrate concentration in all wells, groundwater level contours were estimated from a geodetic survey of the water levels in selected wells. The flow direction of the groundwater underneath the study area was determined using these contours.

Results and Discussion

East Java Surveys

Of the 574 wells surveyed during the dry season in East Java Province, 17% had a NO_3-N concentration above the maximum level of 10 mg/L recommended by the WHO, while for the 551 wells surveyed during the wet season in the same area, 28% were above this level (Table 1).

Table 1. Percentage distribution of number of wells within specified NO_3-N concentration ranges for the surveys in the dry season 1991 (n=574) and wet season February 1992 (n=551).

NO_3-N range (mg/L)	1991 %	1992 %
0–5	60	51
5.1–10	23	21
10.1–15	8	11
15.1–20	2	5
>20	7	12

In the same province, there was great variation in NO_3-N concentrations between regions and settlements (towns and villages) independent of the season. For instance, in the Jombang region there were more wells with a NO_3-N concentration above 10 mg/L than in the Madiun region. The Madiun region is located in an area of heavy clay soils while the Jombang region soils are predominantly well-drained volcanic types. In general, regions with soils with an expected low hydraulic conductivity (such as heavy clays) were invariably low in NO_3-N. This is perhaps due to temporary water ponding on the heavy soil during rainy periods, creating anaerobic conditions, which would be conducive to denitrification.

Not only was there variation in the NO_3-N concentrations between settlements, but also within them (Table 2). Wells with NO_3-N levels far above 100 mg/L were sometimes located less than 30 m from those with less than 10 mg/L.

Table 2. Percentage distribution of the number of wells within a specified NO_3-N concentration range in the two study villages Mojosari (n=51) East Java, and Kota Gede (n=113) Central Java sampled in the wet seasons 1992 and 1994 respectively.

NO_3-N concentration (mg/L)	Mojosari % of wells	Kota Gede % of wells
0–5	14	3
5.1–10	39	17
10.1–20	29	62
20.1–30	14	11
30.1–40	2	5
40.1–50	0	1
50.1–60	2	0
>60	0	1

The results of the surveys suggest that millions of people in Java regularly consume water with a NO_3-N concentration above the WHO recommended standard of 10 mg/L. Probably millions drink water with a NO_3-N level greater than 22 mg/L. These estimates do not take into account the fact that virtually all households using well water for consumption boil it to eliminate bacterial contamination. The longer the water is boiled the higher the nitrate concentration becomes, due to evaporative water loss. A survey of some households in Kota Gede indicates that the nitrate concentration can be increased in this way by up to 44% (Table 3).

Furthermore, it is likely that more nitrate is ingested when eating rice—the rice containing all the nitrate of the water in which it was boiled. To this must be added nitrate when eating green, leafy vegetables such as *kangkung* and *bayam*, which can accumulate large amounts of nitrate. Thus any assessment of the impact of nitrate intake on population health should not be based solely on well water surveys.

Table 3. NO_3-N concentrations in well water and boiled water taken from the same well for seven households in Kota Gede.

Water	Household							
	A	B	C	D	E	F	G	Mean
	NO_3-N (mg/L)							
Unboiled	8	18	28	32	33	35	68	
Boiled	10	26	29	39	40	44	71	
Increase (%)	25	44	4	22	21	26	4	21

Origins of Nitrate

Most villages surveyed in east Java were surrounded by rice fields, and therefore, if the nitrate originated from N fertilisers, the NO_3-N concentration in the wells close to the edge of the village would be higher than in wells closer to the centre. Almost without exception the opposite was found. In addition, NO_3-N concentrations in *sawah* floodwater, groundwater directly underneath rice fields, irrigation canals, and well water at the edge of rice fields, were invariably between 0 and 3.4 mg/L (Wetselaar et al. 1993). These findings are not surprising considering that nitrate is very unstable under the prevailing anaerobic conditions of a flooded rice field (Patrick and Reddy 1977; Simpson et al. 1984). Moreover, any leaching will be slow in view of a generally low hydraulic conductivity of the soil due to a hard compacted layer at about 15 cm produced by puddling.

Other possible sources of nitrate considered by Wetselaar et al. (1993) are rainwater, dinitrogen fixation, soil organic N, rubbish pits, sewage and animal manure. The amount of the first two were too small to account for the high nitrate concentrations found in wells. Soil organic N is generally concentrated in the top 15 cm of a soil profile. When wet, for instance, due to rainfall, a part of this N can be nitrified and nitrate leached down the soil profile into the groundwater. Such leaching can only occur where the nitrate is not taken up by trees or plants growing in the soil, and therefore, may be significant in bare soils. In many parts of the villages surveyed, there were bare soil areas. When these were sampled it was found that the amount of nitrate in the soil profiles varied greatly within and between settlements (Fig. 1) but in each case the organic N content in the topsoil was invariably low. In addition, the form of this organic N was such that it had a low capacity to be mineralised. For example, when 18 topsoils from Kota Gede were incubated anaerobically for one week, only 0.22% ± 0.10% of the organic N was mineralised compared with 2.02% for a sawah soil adjacent to the settlement. Thus in village soils in which appreciable nitrate levels were found in the profile, the nitrate must have originated from an organic source, which in the process of mineralisation, had largely been depleted. Although soil organic N in the bare areas of villages

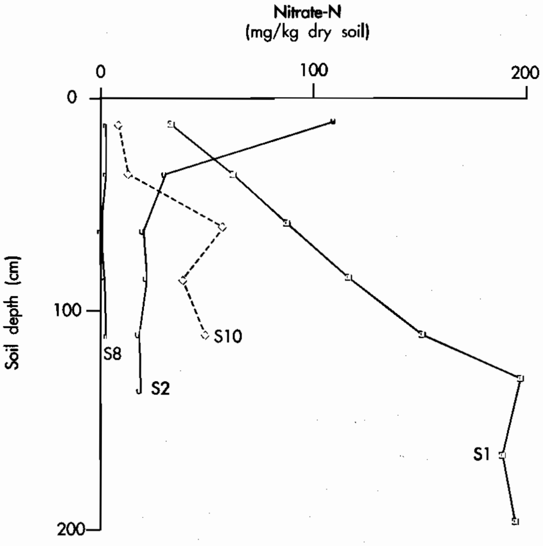

Figure 1. Nitrate-N concentrations in different soil layers of four cores collected from bare soil areas. S1 and S2 were located in one village, and S8 and S10 in another (Wetselaar et al. 1993).

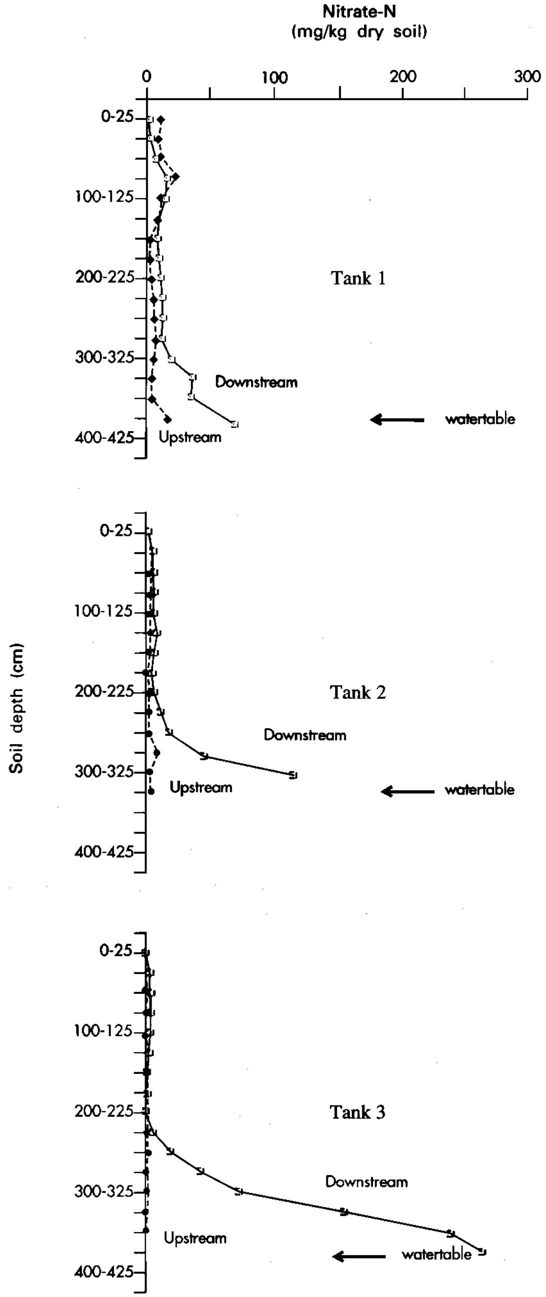

Figure 2. Nitrate-N concentrations in different soil layers in cores located either 1 m upstream or 1 m downstream of three septic tanks, the stream flow relating to the groundwater. The horizontal arrow indicates the approximate groundwater depth at time of soil sampling.

can generate nitrate, the capacity to do so decreases with time. Therefore, it is unlikely that the major part of the nitrate observed in many wells originated mainly from bare soils inside settlements unless this nitrate is not of recent origin.

Nearly all protein consumed by humans in settlements is brought in from the surrounding fields. Thus, the N contained in this protein adds to the N load from which nitrate can be generated after ingestion and excretion. In the absence of sewage systems such as toilets with septic tanks, some of the excrement is deposited in designated areas in the village. Very conservative estimates by Wetselaar et al. (1993) indicate that average NO_3-N concentrations of over 68 mg/L can be generated annually in the water in the soil profile below a designated area. Where septic tanks are used, a higher NO_3-N level can occur due to a high, local concentration of the source. NO_3-N levels of up to 950 mg/L can be expected in the soil water in the immediate vicinity of the septic tank outlet.

Soil sampling in Kota Gede, about 1 m away from three septic tanks, suggests that the estimated NO_3-N concentration near the tanks is close to reality. In Figure 2 results for the three tanks are given for soil cores taken either upstream of the tank (in relation to the groundwater flow) or directly downstream. In this figure, NO_3-N is expressed in mg/kg dry soil, so that at a moisture content of approximately 50%, the NO_3-N concentration in the soil water will be twice that of the NO_3-N concentration for the soil.

The nitrate concentrations measured in the soil just above the groundwater table showed enormous differences between upstream and downstream cores (Fig. 2) even though these were only a few metres apart. This clearly demonstrates that septic tanks can be major point sources for nitrate plumes downstream of the tanks. It is possible that the water in any well placed directly downstream of septic tanks would be highly contaminated with nitrate.

In Kota Gede virtually all households have septic tanks and wells, with the location of the wells seemingly placed at random in relation to possible nitrate plume from the tanks. As could be expected, random distribution of high and low well water nitrate concentrations was evident over the surveyed areas. In contrast, the Mojosari area with high nitrate concentrations was confined to a small zone in the centre of the village which coincided with the only area where households have septic tanks.

Rubbish pits, locally known as '*lubang sampah*', could also act as point sources for nitrate. These pits are 2 m^2 holes in the ground and are about 2 m deep. Almost every household has one of these in its yard at a given time, and is used to discard rubbish such as leaves and food waste. In general, one would expect the C:N ratio of the material in the pit to be high, and consequently, little nitrate would be generated. Soil sampling in Kota Gede beneath a pit that was in-filled one year before, showed that the NO_3-N level in the soil below the original pit was not significantly different from a control soil core 2 m away from the pit (Fig. 3).

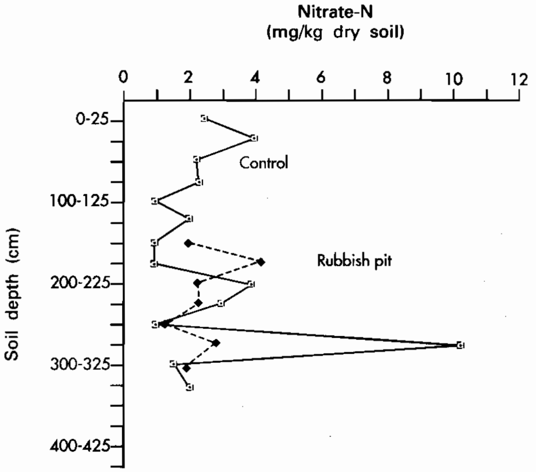

Figure 3. Nitrate-N concentrations in different soil layers in cores located either underneath a rubbish pit or 3 m upstream of it (control) the stream flow relating to the groundwater. The horizontal arrow indicates the approximate groundwater depth at time of soil sampling.

Overall, the human excretion of N within a settlement can have a profound effect on the quality of drinking water. A lesser effect arises from animals kept in a village. The average cattle density in rural east Java is one animal per three families (Wetselaar et al. 1993) and there is an increasing use of small battery stalls for chickens. The concentration of cattle in yards can lead to the build up of strong point sources for nitrate. High nitrate levels were indeed found in the soil below current and old yards.

Conclusions

The results of the surveys in East Java suggest that perhaps up to one quarter of the population drink water with a NO_3-N concentration above the maximum level recommended by the WHO. Where the water table is below 2 m depth, the occurrence of well water with high nitrate levels is unpredictable at this stage, but it appears unlikely that the main cause for this occurrence is the application of N fertilisers to lowland rice fields. Human excretion of digested food is the most likely origin of the nitrate found in groundwater underneath settlements. With septic tanks, this source becomes highly concentrated and gives rise to nitrate plumes. Any wells located within these plumes can expect nitrate-contaminated drinking water.

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Nutrient Leaching Beneath Horticultural Land Use — West Australian Examples

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Abstract

This paper briefly describes the use of a field technique, using non-weighing lysimeters, to simultaneously measure the water and solute fluxes below the root zone. Nutrient leaching was studied using the technique in horticultural land use in coastal sandy soils in Western Australia. Heavy fertilising and over-irrigation practised in the industry has led to substantial leaching loss of N, causing serious contamination of underlying aquifers. In certain soils substantial P leaching has also occurred, which has been responsible for a serious eutrophication problem in the region. Drainage below the root zone accounted for up to 50% of the total water input and contained about 40% of the applied N (mainly as NO_3-N) and about 12% of the applied P. The results point to the large scope for modifying management practices for safeguarding the quality of water resources, and the more efficient use of fertilisers.

INCREASING use of agrochemicals to enhance food production has occurred around the world over the last several decades. Horticulture, an intensive form of agriculture, relies on heavy use of fertilisers, pesticides and irrigation, so that crops of high value and quality are produced. However, such management practices can lead to excess discharge of nutrients N and P, as well as pesticides, to water resources. The intensity of groundwater pollution through agrochemicals is likely to be high under light textured soils, which not only have high recharge rates, but also low adsorption capacities for agrochemicals.

N and P are the prime nutrient pollutants. From a human health view point, nitrate is of primary concern. The suggested World Health Organisation (WHO) limit of NO_3-N in drinking water is 10 mg/L. Enrichment of surface waters with N and P promotes eutrophication. Usually P is the key limiting element (Vollenweider 1968) its environmental limit being 0.025 mg/L to 0.05 mg/L, while for N it is 0.05 to 1 mg/L (Ryding and Rast 1989; Chapman 1992). Serious problems through nitrate contamination of groundwater have been reported around the world (e.g. Strebel et al. 1989; Goodrich et al. 1991; Spalding and Exner 1993; Sharma 1994).

The sustainability of intensive agriculture will depend on its environmental compatibility. Accurate assessment is needed of environmental consequences of the industry, so that management practices can be modified to minimise the damage.

The objective of this paper is to describe briefly the experimental technique developed to quantify dynamics of N and P leaching under horticultural land use. Results obtained from studies of West Australian sandy soils will be discussed.

The Study Area

The Swan coastal plain is an important agricultural area on the Western Australian coast. It surrounds the capital city of Perth (Fig. 1) which has a population of over one million. It is a major source of groundwater, and over 40% of the city's water supply is derived from it. Groundwater is also a major source of water for the Swan-Canning, and Peel-Harvey estuaries and many shallow lakes. Nutrient contamination of groundwaters as well as many water bodies is a major concern.

Because of the proximity to the population centre and easy access to shallow aquifers for irrigation, a sizeable part of the Swan coastal plain has been developed for horticulture. Irrigated horticulture includes market gardens, orchards, vineyards and nurseries, and is considered a major nutrient source, as groundwaters have been found to be contaminated especially with nitrates (Poinke et al. 1990; Gerritse et al. 1990; Sharma et al. 1991b).

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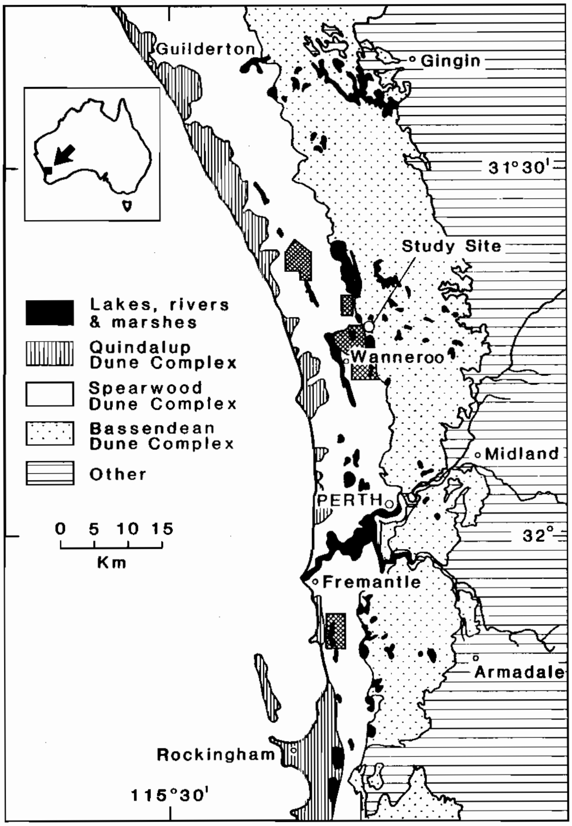

Figure 1. General morphology and land use of the Swan Coastal Plain, Western Australia, and location of experimental site.

The area has a Mediterranean climate with distinct summer and winter seasons. Annual precipitation of about 800 mm is the primary source of natural recharge. Potential evaporation far exceeds rainfall; class A pan evaporation is about 1800 mm/yr. Irrigation water for horticulture is derived from the underlying shallow unconfined aquifer, and is about 1500 mm/yr.

Leaching Potential

Estimates of maximum potential nutrient losses through leaching can be computed through mass balance of nutrients in the root zone. For example, water balance for a land unit can be expressed as:

$$p + I - ET - RO - \Delta S - R = 0 \qquad (1)$$

where p is precipitation, I is irrigation, ET is evapotranspiration, RO is runoff, ΔS is increase in soil water storage in the root zone, and R is percolate below the root zone (≈ drainage or recharge).

Similarly, the corresponding mass balance for a nutrient can be expressed as:

$$p' + I' + F + M - RO' - \Delta S' - C - DV - R' = 0 \qquad (2)$$

where p', I', RO' are nutrient input/output in precipitation, irrigation and runoff respectively, ΔS' is increase in nutrient storage in soil within the root zone, F and M are nutrient added in fertiliser and manure respectively, C is nutrient removed by crop harvest, DV is nutrient lost through volatilisation and other biochemical processes (e.g. denitrification) and R is nutrient leached in percolate.

When computing leaching, the most significant hydrological term required is recharge. For sandy soils with low slopes, surface runoff is negligible. Both p and I can be measured accurately, ΔS can also be measured reasonably accurately over a period of a week or more. Over longer time intervals (i.e. year), ΔS can be assumed negligible. If ET was known, recharge can be calculated by the difference.

For computing nutrient balance, the major input terms (i.e. manures and fertilisers) need to be quantified accurately. Contributions from precipitation can usually be ignored. However, groundwater used for irrigation may contain reasonably high nutrient levels (especially N) and this term needs to be quantified. For example, 1000 mm of irrigation water on the coastal plain could contain up to 560 kg/ha N, with a median of about 200 kg/ha N (Poinke et al. 1990).

The storage term for nutrients can be very large in relation to the other components. Most cultivated sands contain 1–2% organic carbon (OC) in the top 50 cm interval. Assuming ratios C:N and C:P of 1:10 and 1:100, the storage terms for the top 50 cm of soil for each 1% OC will be 8500 kg/ha N and 850 kg/ha P. Furthermore, P adsorption in the finer mineral fractions will be about 255 kg/ha, thus giving a total P storage of about 1100 kg/ha.

The dominant sources of nutrients on the Swan coastal plain are poultry manure and commercial fertilisers. An average crop application is estimated to be about 1000 kg N and 290 kg P. For a maximum of three crops grown annually, the total input would be 3000 kg/yr N and 870 kg/yr P. Nutrient export in the crop harvest amounts to 10–15% of the input, thus leaving an excess of about 2700 kg/yr N and 780 kg/yr P. An additional N input in 1000 mm of irrigation water would amount to about 200 kg/yr N. Assuming an annual total water input of 1800 mm/yr (1000 mm irrigation + 800 mm rainfall) and assuming leaching of all the excess nutrients, then the estimated potential NO_3-N and P concentrations in the percolate would be about 360 mg/L N and 100 mg/L P. The above computations assume all excess N and P are transformed to nitrate and phosphate, with no change in storage and no losses of N through volatilisation/ denitrification.

Loss of nutrients through processes such as adsorption, volatilisation and denitrification, means the actual concentrations of N and P in the leachate below the root zone and in the groundwater are likely to be less than the potentials. The relationship between the potential and actual is complex, and will depend on the properties of the system (e.g. soil properties, management conditions, type of crop, temperature, water availability, and aeration condition).

A survey of groundwater wells beneath two horticultural areas on the Swan coastal plain showed that the average NO_3-N concentrations were 31 mg/L and 10 mg/L, while the NH_4-N concentrations were <0.8 mg/L. The N export to groundwater represented <10% of the estimated leaching potential. The P concentrations were low and highly variable (0.01 to 0.09 mg/L) with an average of 0.03 mg/L, representing <<0.1% of the leaching potential (Poinke et al. 1989; Sharma et al. 1991b).

Translation of leaching potential to actual in groundwater depends on many physical, chemical, and biological factors. This requires understanding of water and pollutant retention, transformation, and transport mechanisms. The translation of leaching potential to actual can be greatly assisted with the aid of modelling, which must incorporate the key mechanisms and major influencing factors. This can be further assisted by the actual field measurements.

Evaluation of detailed dynamics of water/nutrient fluxes in the field is difficult, if not impractical. It has been shown that water/nutrient flux measurements at a reference depth below the root zone will provide useful data against which models can be calibrated/verified, and if necessary modified.

Figure 2 shows the concept of potential, reference, and actual pollution in relation to nutrient leaching and groundwater contamination.

Field Measurement of Reference Water/Pollutant Fluxes

To accurately measure the water and nutrient fluxes below the root zone, non-weighing lysimeters were designed (Sharma et al. 1989; 1991b). Measurements were made at two market garden farms representative of the Swan coastal area.

The general design of the lysimeter is shown in Figure 3. The details of installation and validity of measurements are discussed in Sharma et al. (1989). The lysimeter was constructed in two parts. The upper cylinder consisted of a 1.5 m length, with a 1.5 m diameter, and was made from 1.2 mm galvanised iron. The base tank of similar diameter had a slight slope, was made from mild steel, and galvanised. A water control housing unit was attached to the base tank, and with the aid of a pump enabled pumping of leachate. The upper cylinder was joined to the base tank, but separated by galvanised wire mesh and geofabric cloth. This provided free drainage to the base tank, from where leachate could be pumped out as desired. The lysimeters were installed by backfilling the layers of sand and soil carefully in the same order and bulk density as originally found in the field. The top of the lysimeter tank was 0.6 m below the soil surface.

The lysimeters were positioned, as far as possible, beneath the representative landscape areas of the two farms. Hydrological parameters, (irrigation and rainfall), soil water storage, fertiliser, and manure inputs were measured. The crop yields were also measured and nutrient compositions of plant parts were analysed.

Nutrient Leaching

A comparison of soil water storage in the top 2 m of the soil profile inside and outside the lysimeters (Fig. 4) shows that the hydrological behaviour inside the lysimeters was very similar to the undisturbed field. This gives confidence in the lysimeter data.

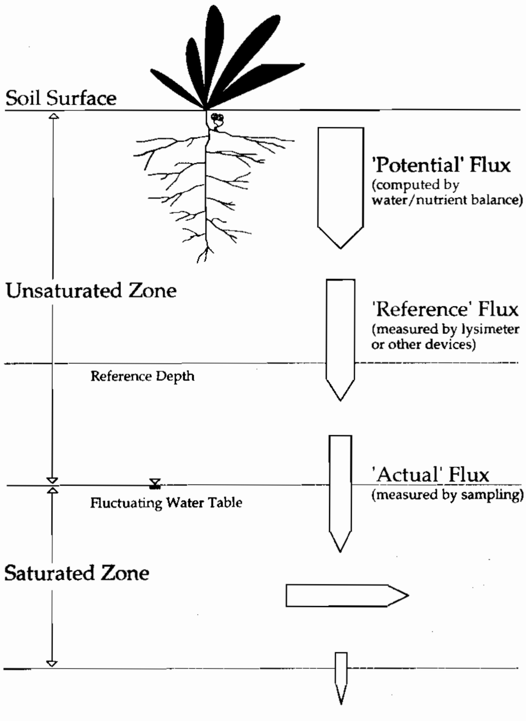

Figure 2. Schematic diagram showing relationship between Potential, Reference and Actual leaching concepts in relation to groundwater contamination.

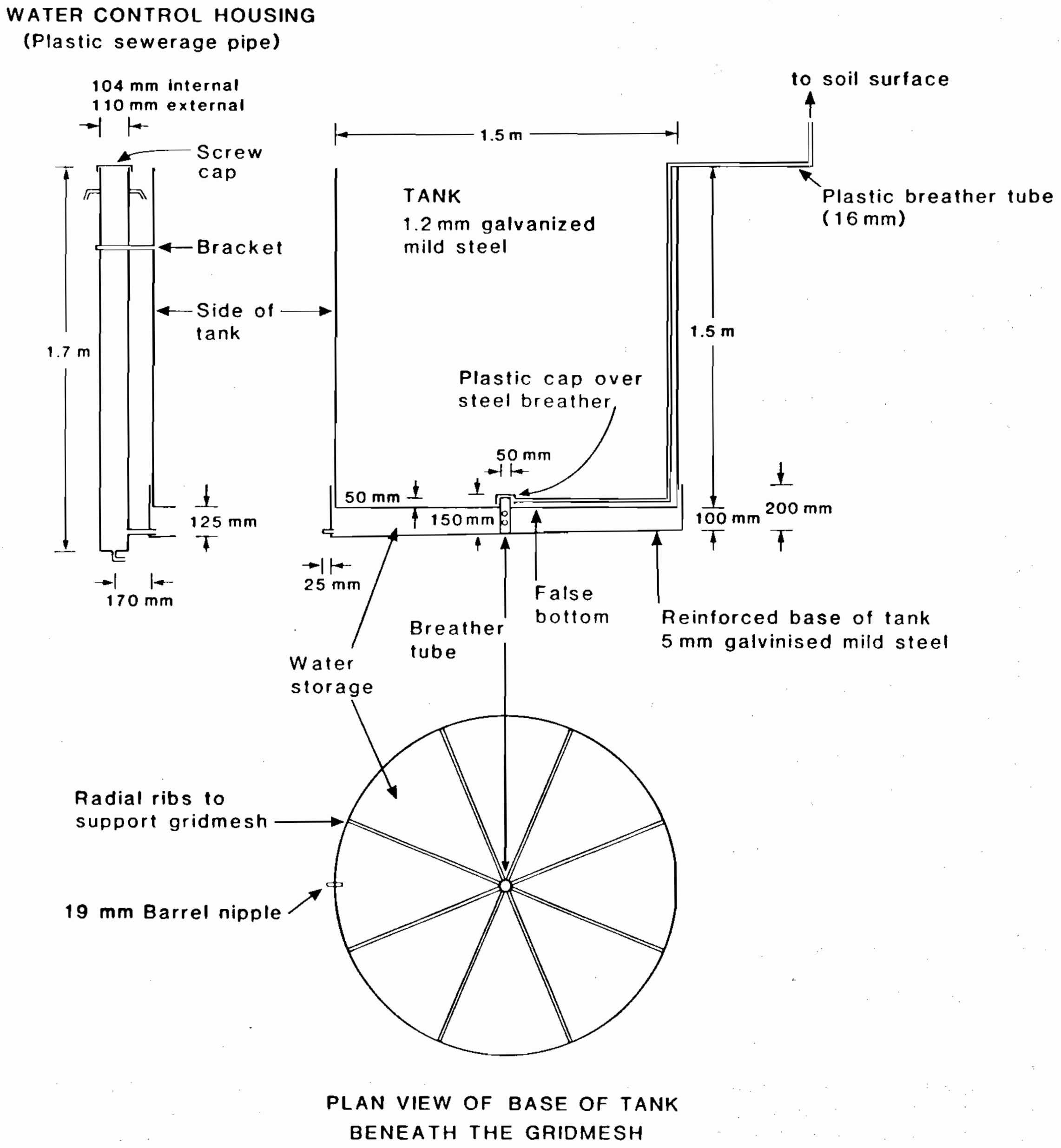

Figure 3. Design of the lysimeter and positioning of the water control housing unit.

Average cumulative values (over three lysimeters) of the components of hydrological balance i.e. total water input (I_T = irrigation + precipitation), drainage below the root zone (R) and evapotranspiration (ET) for a farm from January 1989 to July 1991 are presented in Figure 5. The evapotranspiration was calculated for each period (weekly/fortnightly) of measurement from:

$$ET = I_T - \Delta W - R \qquad (3)$$

where W is the change in soil water storage. The values of some of the ratios and cumulative parameters are presented in Table 1. These are for a 784 day period covering a cropping sequence of cauliflowers-fallow-potatoes. The overall data show that drainage and evapotranspiration were similar (about 50% of the total water input) and irrigation accounted for about 60% of water input. During summer, irrigation accounted for over 95% of I_T. Although there were large seasonal variations, the total ET was only 54% of Class A pan evaporation (Ep).

Table 1. Average cumulative hydrological parameters and their ratios for a horticultural land use.

Hydrological Parameter		21/03/89 to 29/05/91 (mm)
Precipitation	P	1587
Irrigation	I	2360
Total Input	I_T=P+I	3947
Recharge	R	1923
Evapotranspiration	ET	1948
Open Pan Evaporation	E_p	3584
	ET/E_p	0.54
	R/I_T	0.49
	P/I_T	0.41
	I/I_T	0.61
	ET/I_T	0.50

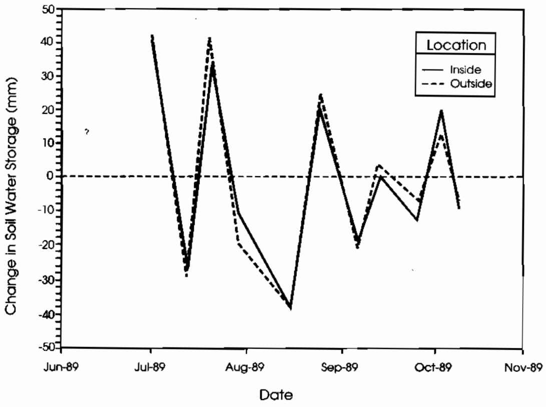

Figure 4. Comparison of change in average soil water storage in the top 2 m of the profile inside and outside the lysimeters.

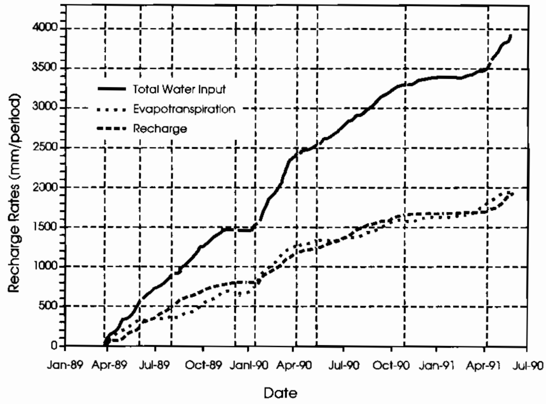

Figure 5. Average cumulative recharge, total water input and evapotranspiration on a horticultural farm.

It was found that most (>95%) of the N was leached in the nitrate form. Time variations in NO_3-N concentrations measured in three replicate lysimeters are presented in Figure 6 and demonstrate the effect of spatial variability. It was found that this was mainly caused by variations in distribution of irrigation input between the three lysimeters (Sharma et al. 1991b). The effects of irrigation distribution observed for P concentrations were similar, but of a much lower magnitude. Components of nutrient inputs and outputs are presented in Table 2.

A combined effect of soil type and management practices on the average NO_3-N and P concentrations of leachate are shown for two farms in Figure 7 — one under Bassendean sand, and the other under Spearwood sand. It is obvious that the P peak has broken through Bassendean sand, while the majority of P was still being absorbed by Spearwood sand. These conclusions were confirmed by P storage profiles (Sharma 1991). Very low retention capacity for Bassendean sand has been demonstrated (Gerritse et al. 1990; Sharma et al. 1991a).

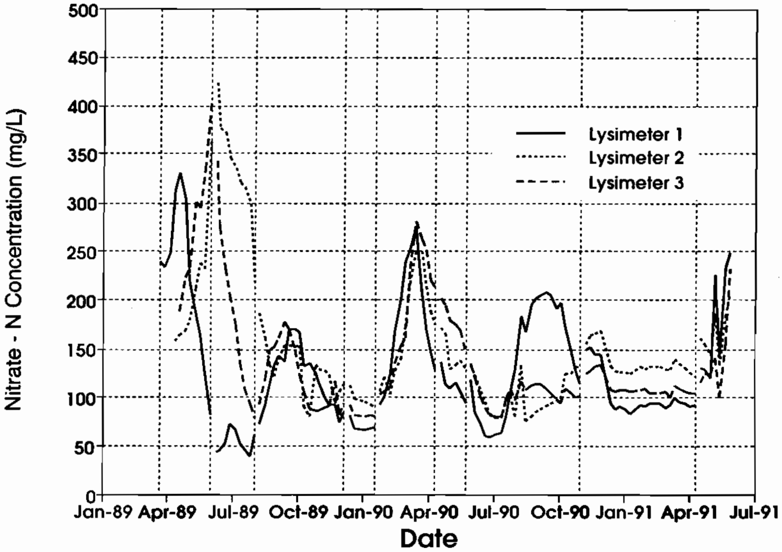

Figure 6. NO_3-N concentrations in leachates collected by three lysimeters beneath a horticultural farm.

Table 2. Nutrient input and output components for a horticultural land use.

Input	Nitrogen kg/ha	% of total	Phosphorus kg/ha	% of total
Manure	2565	55	935	59
Fertilisers	1396	30	648	41
Irrigation	663	15	< 2.1	
Rainfall	2	<< 1	< 1	
TOTAL	4626	100	1584	100
Output	**kg/ha**	**%**	**kg/ha**	**%**
Leachate	1813	39	184	12
Crop Harvest	760	17	128	8
Unaccounted for	2053	44	1274	80

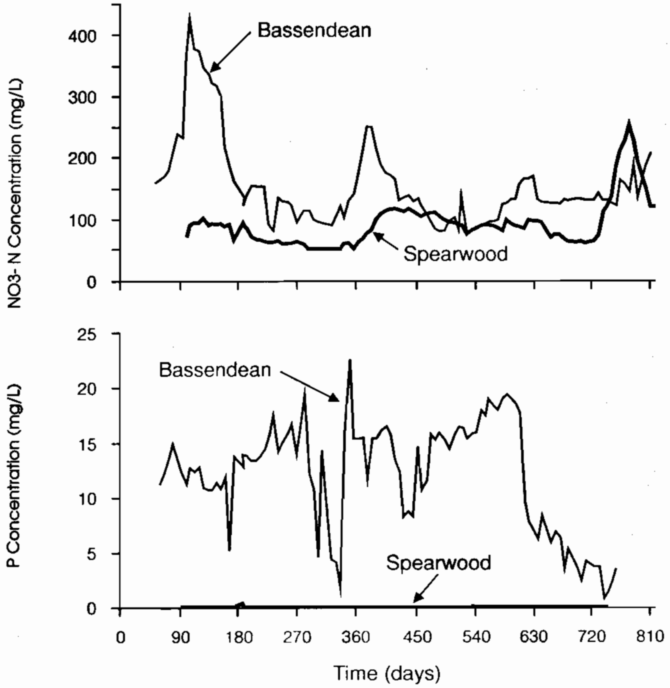

Figure 7. Average NO_3-N and PO_4-P concentrations in leachate beneath horticultural land use on two sandy soils.

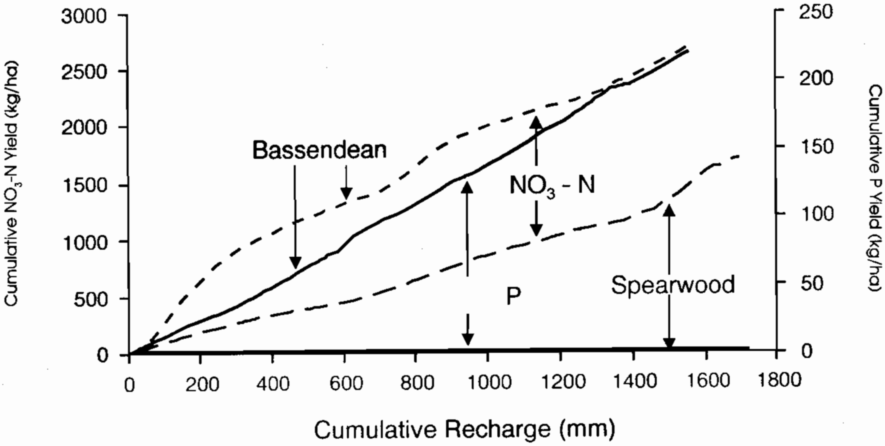

Figure 8. Cumulative leaching yield of NO_3-N and P as a function of cumulative recharge beneath horticultural land use on two sandy soils.

Serious eutrophication problems in lakes and estuaries observed in the coastal region of Western Australia are ascribed to discharge of high levels of P primarily from Bassendean sand (Gerritse and Schofield 1989; Sharma et al. 1993).

Relationships between cumulative yields of NO_3-N and P and the cumulative drainage for the two horticultural farms (Fig. 8) are almost linear. This implies that high nutrient contaminations are associated with high discharge rates (Fig. 6). Obviously the Bassendean system discharges much higher loads of nutrients, specifically P.

Summary and Conclusions

This paper briefly describes a technique based on the deployment of non-weighing lysimeters, and demonstrates its application in intensive agriculture in Western Australia. Such information not only provides insight into the mechanisms of leaching, but can serve as a measurement reference against which alternative measurement techniques can be compared, and simulation models can be calibrated and validated.

Based on measurements carried out over a period of about 2 years from the horticultural farms in Western Australia, the following conclusions can be made.

- Due to heavy fertilising and over-irrigation, up to 50% of the total water application ended up as drainage water, and about 40% of the applied N was leached below the root zone.
- The NO_3-N concentrations in drainage water far exceeded the WHO recommended limit all year round and reached values of up to 400 mg/L.
- Less than 20% of the applied N and about 10% of the applied P was taken up by plants. There was only a minor effect of soil type on N leaching measured.
- On one sandy soil, the measured P concentration in the leachate was very high (5–25 mg/L) while on the other sandy soil, concentrations were relatively low (<0.4 mg/L) due to P adsorption by the soil. The P leaching was restricted to <12% of the total applied.

The data presented in this study demonstrates that there is scope to modify irrigation and nutrient management practices, and as a result, contamination of groundwater and surface water can be reduced.

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Potential Pollution of Surface and Ground Waters in Wetland Rice and Vegetable Production Areas in Indonesia

Naik Sinukaban

Abstract

Agricultural development associated with the intensification of wetland rice and vegetable production in Indonesia may lead to the potential to pollute ground and surface waters. Application rates of fertilisers and pesticides are relatively high. There is little data on the concentrations of N and P and pesticides in surface water of sawah and irrigation channels (which are also used for drinking water). There are indications that nitrate and aldrin occur in excessive concentrations. Nutrient concentrations in a creek of a small catchment were relatively high and eutrophication has been observed. There is little data for groundwater. Given that there is high use of agrochemicals and a large rural population, systematic research should be undertaken to assess the extent of water contamination.

AGRICULTURAL development associated with the intensification program in wetland rice and vegetable production areas has the potential to pollute ground and surface water in Indonesia.

The intensification program is the most popular agricultural development program used to increase rice production in Indonesia. This program aims to maintain self sufficiency in rice and food production. One of the key factors to the program's success is the use of fertilisers and pesticides.

In 1992 the total area of wetland rice fields in Indonesia was almost 11 million ha. The types of fertiliser used for wetland rice are: urea, triple superphosphate (TSP) and potassium chloride (KCl). They are usually applied at a high rate. Pesticides are also used at a high rate to control insect pests and diseases. Planting occurs at least twice a year in wetland rice and vegetable production areas. The average rice yield in this program is 4.97 ton/ha.

In 1992 vegetable production areas in Indonesia covered 312 500 ha (Statistical Year Book of Indonesia 1992). The rate of fertiliser and pesticide use on vegetable production areas is higher than in the wetland rice fields. The types of fertilisers used in vegetable production areas are urea, TSP, KCl, and manure. The pesticides used are Dicarzol and Ambush, and the fungicides are Vandozef and Polyram. The effects of these chemicals on groundwater and surface water pollution are not yet known. Research on chemical contamination is very limited.

Potential of the Water Quality Problem

Wetland rice field areas in Indonesia during the Fifth Five Year Plan covered more than 10 million ha each year, with an increase of 2.1% per year (Table 1). At least 20 million farm households are involved in this program.

Table 1. Wetland Rice Field Areas in Indonesia During the Fifth Five Year Plan (1000 ha) *.

Program	Fifth Five Year Plan			
	1989	1990	1991	1992
Intensification	8826	8876	8734	9281
Non Intensification	1705	1626	1548	1679
Total	10 531	10 502	10 282	10 976
Area:				
Java	5448	5419	5184	5553
Outside Java	5083	5083	5098	5425
Total	10 531	10 502	10 282	10 978

* Source: The President of the Republic of Indonesia Speech before the House of Representatives, 1 March 1993.

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Soils in the wetland rice area are generally clayey (Grummosols, Alluvial soils) although in some areas (Klaten, Central Java) rice is planted in sandy soils. The rate of fertiliser application for rice in the intensification program is 250–350 kg/ha urea, 150 kg/ha TSP, and 100 kg/ha KCl per planting. The application of pesticides for rice ranges from 3 to 6 kg/ha/planting. Planting occurs twice each year. Therefore, the amount of chemicals used per year is twice the per ha amount. The excessive use of chemical fertilisers and pesticides may contaminate the soils, surface water and groundwater.

Table 2. Concentration of nitrate and pesticides in water and soil in the several rice production centres of West Java and Central Java.

	Sample Type	Conc. of NO_3-N (mg/L)	Concentration of Aldrin	
			water (mg/L)	soils (mg/kg)
West Java				
1. Sinalangka, Sukatani	IC	7.8*	-	-
	SW1	10.7	-	-
	SW2	10.4	-	-
2. Bagan, Karawang	SW1	12.7	-	-
	SW2	8.8	-	-
3. Margahayu, Bandung	IC	4.5	-	-
	SW1	2.8	-	-
	SW2	2.8	-	-
4. Pegaden	IC	5.8	-	
	IC	4.8	-	-
	SW1	1.8	-	-
Central Java				
1. Tegal, Pasurungan	SW	0.24	0.0012**	0.0017**
2. Pemalang, Petarukan	SW	0.12	0.0014	0.0026
	IC	-	0.0014	-
3. Pekalongan, Rembun	SW	-	0.0019	-
Krasak	SW	-	0.0014	-
4. Batang, Karanganyar	SW	0.17	0.0016	0.0031
Sambong	IC	-	0.0016	0.0044
5. Sukaharjo, Mulur	IC	0.06	0.0016	0.0044
6. Karanganyar	SW	0.45	-	0.0034
7. Boyolali, Batur	SW	0.10	-	0.0045
8. Klaten, Kepanjen	SW	-	0.0024	-
9. Yogyakarta, Balecatur	SW	0.31	0.0024	0.0016
10. Magelang, Sriwadari	SW	0.38	0.0029	0.0017

*	Critical level of NO_3-N in water is 10 mg/L
**	Critical level of Aldrin in water is 0.001 mg/L
-	No data available
IC =	Irrigation channels
SW =	Surface water in sawah (paddy field)

A survey carried out in 1994 shows that there is nitrate contamination, and pesticide residues in the surface water and soils (Table 2). Water samples were taken from rice fields and irrigation channels, and soil samples were collected from rice fields in the rice production areas along northern coast of West Java and Central Java.

It is clear from Table 2 that some of the irrigation water in channels and surface water have NO_3-N and pesticide concentrations higher than their critical levels for drinking water. In the rural areas, most of the farmers use the water from irrigation channels for drinking and other households purposes, and as a result, this contamination may cause a problem. Data on the effects of chemical use on groundwater quality are not yet available.

The known effects of excessive chemical use in vegetable production areas on surface and groundwater quality is also very limited. Research from Pangalengan, West Java which is one of the centres for vegetable production, shows that the contamination of N and P in surface water from the vegetable production areas (Table 3). This contamination may contribute significantly toward eutrophication of rivers, lakes, and dams.

Table 3. Concentrations of N and P in a creek of a small catchment in Pangalengan, West Java.

Month	NO_3-N mg/L	NH_4-N mg/L	Dissolved P mg/L
November	8.2	0.55	0.06
January	0.15	0.34	0.03
February	0.4	0.57	0.02
March	4.69	0.94	0.03
April	2.53	1.46	0.09

Importance of the Problem

The extent of the contamination problem is not yet known. Data available for the chemical contamination of surface water or groundwater are very limited. However, the data in Table 2 indicates that some surface water in the rice field environment have shown levels of NO_3-N and pesticides higher than critical levels. Most of the farmers get drinking water from irrigation channel/ surface water. The population of farmers involved in rice field areas is about 10 million in Java alone, consequently, the contamination problem might be very serious in the future.

In addition to the contamination of surface water, the sediment and nutrient loss (N and P) through erosion from the rice fields and vegetable production areas may also cause eutrophication in rivers, dams, and lakes. These problems have been observed in Rawa Pening Lake in Central Java and Kerinci Lake in Jambi. These two lakes have extensive infestation of aquatic vegetation, particularly *Echinochloa sp.*

Status of the Problem

The seriousness of the chemical contamination of surface water or groundwater in threatening human life is not yet known due to the lack of available data. However, given the extensive use of chemicals and the large number of farmers involved in this practice, the accumulation of the contaminants might cause very serious problems in the future. Therefore, systematic research on the present status and potential problem of the chemical contamination of surface water and/or groundwater and methods of control are urgently needed.

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Assessment of Agricultural Systems and their Impact on the Quality of Groundwater Resources in the Philippines

Perfecto P. Evangelista

Abstract

This report outlines the problems of increasing agrochemical pollution in the ground and surface water in the rural provinces of Tarlac (low land sugarcane farming) and Benguet (high land vegetable farming) in the Philippines. There are periodic reports of biological and chemical pollutants rendering aquifers non-potable, eutrophication of rivers and instances of poisoning from polluted water. The paper explores possible areas of research, the capability and capacity to undertake a research program to address the problem of agrochemical pollution in the Philippines, and the need for outside technical assistance.

Potential Water Quality Problems from Use of Agrochemicals

THE USE of agrochemicals (pesticides, fungicides, herbicides and fertilisers) is fundamental to the production of the major agricultural crops in the Philippines including rice, corn, vegetables, sugarcane, pineapple and tobacco .

The drive for increased crop production for domestic consumption and export in recent years, has resulted in the extensive use of agrochemicals. Regrettably, exact information on the amount of agrochemicals used on the major agricultural crops is not available. An efficient, systematic monitoring system is needed to measure the amount of chemicals and fertilisers used.

Although rice production occupies the largest area in terms of distribution in the Philippines, groundwater pollution resulting from the use of agrochemicals may not be a serious problem, since rice farmers normally do not follow the standard recommendations for application. Typically, half of the actual recommended amount is applied.

Based on data provided by the Philippine Sugar Regulatory Administration Research and Development Department, sugar planters in the country fertilise their crop with 100–450 kg of N in combination with approximately 50–310 kg of P and 30–250 kg/ha of K per cropping season.

Sugarcane is grown on soils which are medium to light textured. In the province of Tarlac, sugarcane is the mainstay of the economy. Leaching and/or persistence of fertiliser residue in the soils is yet to be confirmed. An important factor to consider in the dynamics of the residues within the soil system, is the high rainfall in this area which is 1850 mm/year with a marked seasonality in the rainfall pattern. The potential for leaching of agrochemicals will be high in the wet season when rainfall exceeds evaporation.

Data from the vegetable growing province of Benguet in the Cordillera Administrative Region (CAR) shows that vegetable farmers use approximately 2 kg/ha of pesticides and/or fungicides per crop. As farmers normally plant three to four crops a year approximately 6–8 kg/ha of these chemicals are being applied to the vegetables per year. These chemicals may be leached and find their way to the groundwater. In addition, approximately 50–140 kg of N, P, K inorganic fertiliser combined with 5–12 tonnes of organic fertiliser, mostly chicken manure, is applied annually, per vegetable crop. It should be noted that soils are predominantly medium

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textured. Considering the annual rainfall of 2500 mm/year, it is not yet known whether or not dilution of the agrochemicals would be sufficient to render these chemicals to an acceptable level.

Importance of the Problem

The majority of the communities in the countryside or rural areas in the Philippines get their drinking water from groundwater aquifers and/or dams. In the high lands (Benguet Province) their main source is groundwater using deep wells, and also existing small or big dams located within the upstream area of watersheds. In the low lands (Tarlac Province) their source is primarily groundwater. However, it is not uncommon to see families in these areas drinking surface (river/dam) water which is also used as a source of irrigation.

There are confirmed and unconfirmed reports of biological and chemical inorganic toxins polluting the groundwater aquifers, making it no longer potable in the high lands and low lands. The eutrophication of rivers is already evident, and some rivers and creeks in these areas are already considered biologically dead. Some cases of poisoning have been reported as a result of drinking polluted water.

Status of the problem

The present problem of groundwater/surface water pollution in the country is serious. The tolerable limits of certain organic and inorganic pollutants have been presented to the populace repeatedly, by both private and government institutes. This cannot be a basis for complacency, since it is a well known fact that bioaccumulation of these toxins can affect humans on a short and long term basis. Thus there is an urgent need to understand and characterise the present pollution problem within the major agricultural production areas in the Philippines.

A research and development program needs to be established and should aim specifically at the characterisation of the present status of soil pollution, where research results can be recorded, and serve as a basis for sustainable agricultural practices in crop production.

The dynamics of water pollution in Philippine agriculture are yet to be understood. Agrochemical technology could be monitored to make sure that these products are made less harmful to the environment. Understanding the mechanics or processes involved in soil pollution caused by excess fertiliser use can be the basis for increasing the efficiency of fertiliser by using the least amounts, yet still achieving optimum yields.

Interest of the Country

The Philippine President, Fidel V. Ramos, has recognised the problem of soil pollution and degradation of groundwater aquifers by declaring and launching the Philippine National Integrated Pest Management (IPM) on May 3, 1993, which stresses the sparing use of agrochemicals.

The Department of Agriculture, through the Bureau of Soils and Water Management (BSWM) have already initiated a modest research project. The study will focus on the effect of pesticide and fertiliser residues on various Philippine soils, crop productivity and animal health.

In essence, we need assistance in technical and logistical support, as well as predictive modelling, so that research projects can be much more effective, and have a greater impact on the communities concerned. The ultimate benefit will be far reaching as the information can be useful not only to researchers and farmers, but more importantly it can be used as a basis for policy-making on environmental issues. Furthermore, the development of low input technologies on agricultural practices could make the country's crop production competitive with those of other participating countries under the General Agreement on Tariffs and Trade (GATT).

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