Numerical studies of nitrogen flows under effluent irrigated lawns on islands in the Great Barrier Reef

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Foreword

Island-based tourism is a major industry in the Great Barrier Reef area. While resorts are built on islands, which are part of the Great Barrier Reef World Heritage Area (GBRWHA), the waters surrounding the islands form part of the Great Barrier Reef Marine Park. At present, there are about 27 resorts on islands within the boundaries of the GBRWHA. These islands invariably have fringing coral reefs and, in some cases, seagrass meadows surrounding the island shore. Sewage effluent discharges from these resorts has long been of concern to the Great Barrier Reef Marine Park Authority (GBRMPA).

Specific examples of 'problem' sewage systems in the 1980s included those at Green Island, where almost untreated effluent was released at the edge of the reef flat, and at Hayman Island where secondary treated effluent was discharged close to the fringing reef causing measurable reef degradation. The long-term effects of sewage effluents, particularly the nutrients, nitrogen and phosphorus in the effluent, are reasonably well understood. Excessive nutrients cause reductions in coral growth, modify the calcification process, promote the growth of other organisms at the expense of coral and inhibit coral reproduction and recruitment. Much of our understanding of the effects of sewage discharges on coral reef systems comes from the well-studied Kaneohe Bay, Hawaii, where large-scale sewage discharge over many years to an enclosed bay led to almost complete destruction of the reef system. Diversion of the discharge to the open ocean allowed the reef system to partially recover after some years.

In 1991, GBRMPA introduced new sewage system requirements for effluent discharges to the Marine Park. The purpose of the new requirements was to ensure that sewage discharges did not degrade reefs of the Marine Park. This was accomplished by requiring and encouraging resorts to discharge as little effluent as possible i.e. reuse as much as possible for land irrigation purposes and, if discharging at all, to use tertiary, nutrient reduction treatment of the effluent. Thus, for example, by the early 1990s the systems at Green and Hayman Islands had been upgraded to a high standard. One of the principal strategies of the sewage management policy was encouragement to reuse effluent on the islands for irrigation of gardens, golf courses and other grasslands. GBRMPA needed to confirm that this strategy was effective i.e. that sewage nutrients were stored or eliminated on the land and not returned to reef waters. In addition, an understanding of the pathways of nutrient storage and flow after irrigation was required, especially under the differing geology, soil types, rainfall regimes and vegetation types present on different islands. To gain this knowledge, GBRMPA asked the CRC Reef to undertake a research program, the results of which are presented in this report.

The results of the studies on Great Keppel, Dunk and Brampton Islands explained in this report give GBRMPA, and resort management, confidence that most of the nitrogen applied as effluent to vegetated areas is retained on site in these examples. It is also probable that similar retention of phosphorus occurs although not directly studied in the present work. This knowledge also provides some confidence that other similar resort effluent irrigation schemes may work in an equally satisfactory way although, in the end, each case must be assessed against the circumstances particular to each island and resort. The report is also particularly useful as a resource document for the design and operation of sewage irrigation systems. Recommendations regarding desirable soil depths, irrigation spatial and temporal patterns, windbreaks, land slopes and effluent loadings will be of use to both operators and management for the design and assessment of proposed sewage irrigation systems. Overall this report provides an excellent practical basis for our ability to recommend sewage reuse on Great Barrier Reef islands and minimise marine disposal.

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EXECUTIVE SUMMARY

The influence of dissolved chemicals, especially nutrients such as nitrogen (N) and phosphorus (P), on the water quality of the Great Barrier Reef (GBR) Lagoon is assuming greater importance as pressures for development, both on the adjacent mainland and on islands in the lagoon, increase. Consideration must be given to marine water quality around the main reef structure itself and also around numerous fringing reefs often associated with islands throughout the region. The local marine environment around these fringing reefs will be influenced by discharges from the islands. Given that coral reef ecology is sensitive to relatively small elevations of nutrient concentrations above natural background levels, appropriate management of water and wastewater in resorts on these islands will rely on accurate predictions of the eventual fate of those nutrients.

This report describes numerical studies that have the objective of quantifying the significance of N flows from effluent irrigation practice on resort islands. Our study concentrates on N because, in land disposal of effluent, P will be adsorbed by the soil, reducing its mobility and the environmental risk posed. As effluent irrigation schemes are becoming an increasingly popular alternative to ocean outfalls, their relative success in retarding N leaching was of considerable interest.

In order to estimate upper limits for potential N discharges to the sea, numerical modelling of the fate of N within the unsaturated zone under lawn has been undertaken for three different resorts of climatically and geologically differing characteristics. A series of hypothetical effluent irrigation regimes was considered. These regimes were based on assumed fractions of the actual wastewater produced daily, coupled with data obtained for N measured intermittently in treated sewage over four years.

The parameters required for this study were primarily obtained from literature sources supplemented by some laboratory measurements, with simulations driven by historic meteorological data and experimentally determined soil hydraulic properties.

Sensitivity studies were also undertaken on the effect of spatially variable soil hydraulic properties on the soil water flux model, and parameter variations on the N cycling sub-model.

The first sensitivity study took account of stochastic variations in measured soil hydraulic properties ranging from ± 1 standard deviation from the mean, and defined an envelope of probable outcomes for deep percolation output at each site. A second sensitivity analysis focusing on the effect of changes to the N model inputs highlighted which parameters were most likely to influence the predicted downward movement of N from the soil. An indication of the sign and magnitude of N model output sensitivity was also given for all major parameters.

A broad-based validation process was also conducted to ascertain the applicability and relevance of the simulated values for annual NO_3^- leaching. The predicted soil solution NO_3^- concentration measurements were in reasonable agreement with measured values by the standards commonly found when modelling complex biological systems, and also considering the uncertainty in algorithms and parameter values for a majority of the processes.

The predicted mean annual losses of NO_3^- below the root zone at Great Keppel and Dunk Islands ranged from 30 to 502 kg N yr^{-1} and 28 to 126 kg N yr^{-1} respectively, across four daily wastewater irrigation scenarios applied to the golf course at each resort. Brampton Island, which currently employs effluent irrigation, could expect to generate only between 7 and 38 kg N yr^{-1} flow beyond the root zone of the golf course.

Simulation results for resorts at Dunk and Brampton Islands showed that the transfer of N from the unsaturated zone was reduced substantially for all cases evaluated. Of note however is the significance of rapid transport of N via surface runoff and possible short circuit subsurface flow paths at Dunk Island - intensified by the wet tropical conditions experienced there.

For minor levels of effluent N applied to Great Keppel Island, the soil and vegetation was shown to be reasonably effective in minimising the progress of N to the groundwater system. However, at moderate to high rates of applied N the inherent soil properties and N transformation processes resulted in a more pronounced level of sub-surface N transport. For irrigating large fractions of the total available sewage effluent in this case, it would be advisable to distribute over an area larger than 1.2 ha (golf course) to lower the hydraulic loading.

The effect of recycling turfgrass clippings after cutting was also investigated. A significant addition of N to each island system occurred as a consequence. This diminished the efficiency of N usage by the soil and vegetation in all cases, although N flows from below the profile were not dramatically increased at Brampton and Dunk Islands. Introducing clipping removal practice could further lessen the potential leaching risks from moderate to high wastewater reuse particularly on sandy areas.

For both Dunk and Brampton Islands, the maximum reduction in the potential N to flow to sea was achieved when 100% of the current daily effluent production was distributed over a turfgrass area corresponding to the size of each golf course area. A reduction of 85% and 93% was simulated for each island respectively for such a case. As wastewater application rates rose, steady increases in the simulated reduction of N available for discharge to the sea were predicted.

The reduction in the flow of N below the root zone at Great Keppel Island also reached a peak value for the maximum level of wastewater irrigation loading. The degree of effectiveness of land utilisation of N was not as high here as the other islands studied however, reducing the available N for discharge to sea by 44%. Additionally, relatively small increases in N usage by the land system were associated with much larger increases in the irrigation rate, pointing to a maximum threshold of N uptake by turfgrass being approached or surpassed. It also bears mentioning that as the prospective irrigation areas of each island site were assumed to be no greater in extent than the golf course, larger areas within the resort environs such as airstrips and gardens are likely to be available for use.

In general, predicted outputs should be considered a lower estimate of N reduction in terms of the level of N available for discharge to the local marine environment. This is substantiated by the assumption of N as a conservative (non-interactive) solute entering an aquifer (with no dilution) which freely discharges to the GBR lagoon.

Based on the degree of leaching predicted for Great Keppel Island at high levels of applied N, increases in the nitrate concentration of surrounding waters sufficient to be detrimental to the marine ecology cannot be discounted. Further detailed studies of resort island nutrient mass balances are warranted in conjunction with more detailed work on the groundwater system transporting N from the unsaturated zone to sea.

To complete the prediction of the influence of N export from resort islands on local marine water quality, reliable data on currents and on marine transport processes will be required in the immediate vicinity of the islands. These will govern the actual residence times and control volumes for calculations of local marine nutrient concentration.

From a wastewater management perspective and presuming that high rates of applied effluent irrigation are logistically and economically viable, considerable benefit can be drawn from the high efficiency of N assimilation by the soil-plant-atmosphere continuum in reducing potential N discharge. It is imperative however, that associated health concerns are adequately addressed, which is outside the scope of this study.

The configuration and implementation of an available numerical model for the nutrient cycling in GBR resort island turfgrass systems irrigated with sewage effluent, represents a major step towards a better understanding of the gross outputs expected from such practice.

1. INTRODUCTION

To quantitatively assess the influx of nutrients to the marine microenvironment at GBR resort islands, information is required on the quantity, quality and disposal of effluent, surface and sub-surface nutrient transport mechanisms, and mixing and transport processes in the sea. There are however, substantial gaps in knowledge especially on the storage and transport processes. The terrestrial transport and storage of nutrients will vary with topography, soil type, climatic zone, hydrogeology and vegetation on the islands.

Management practices such as irrigation and fertilisation can also have important ramifications on these conditions. The disposal of wastewater on land is considered a potentially desirable alternative to the discharge of treated sewage into the ocean from a piped outfall. The latter method is still commonly practiced on a number of resort islands within the GBR lagoon. Given that effluent irrigation was a possible preferred method of disposal, further studies were needed to quantify the terrestrial nutrient storage and transport processes since this will determine the location, extent and concentration of discharge to the sea.

In this report, a turfgrass system was considered the most representative ground cover for all effluent irrigation applications. The inherent N levels of most soils are rarely sufficient to meet the nutritional demands of turfgrass; therefore, lawns usually require fertilization with N to maintain a desirable grass quality. Irrigation with secondary sewage effluent can provide large amounts of N to the soil, depending on the concentration of N in the effluent and the volume of water applied. It then follows that this input of nutrients to the soil can be beneficially used by vegetation in the area of disposal. If however, inadequate attention is paid to site characteristics, site management and wastewater quality, runoff or drainage water may carry nutrients from the disposal site, resulting in environmental pollution.

A numerical simulation model, taking the most important processes into consideration, was a valuable tool for evaluating the complex nutrient transformations and interactions. It provided a useful means of predicting the level of nutrients leaching from the unsaturated profile and the changes in the nutrient cycles of the system over time.

1.1 Objectives

The broad aims of this investigation were as follows:

- Synthesize a range of water and wastewater management data pertaining to GBR island resorts;
- Review the current knowledge on nutrient leaching below turfgrass environments;
- Establish representative field monitoring sites based on a range of climatic, hydrogeological and effluent quality and disposal characteristics;
- Apply a numerical nutrient transport model to ascertain the longer term consequences of effluent irrigation applied to turfgrass in coastal systems;
- Perform a sensitivity analysis on selected model parameters to identify the range and magnitude of variations in predicted leaching response;
- Validate predicted modelling outcomes using field data acquired from one or more field stations;
- Present a conservative, broad-based comparison of the mean annual masses of nutrient exported to the GBR lagoon for a variety of effluent irrigation regimes;
- Discuss the utility of the modelling approach and comment on outcomes for management.

Notably, this report focuses exclusively on continental (high) resort islands within the GBR and cannot be easily extrapolated to the hydrogeological and biological conditions found on coral cays such as Green and Heron Islands.

1.2 Significance of Nutrient Enrichment of the Great Barrier Reef Ecosystem

1.2.1 Impact of Nutrient Discharges on Coral Communities

Furnas and Mitchell (1987) described how nutrients were dispersed by water movement and transformed by planktonic biota. Given sufficient inputs of nitrogenous nutrients, phytoplankton could develop into blooms within 2 - 3 days. Although additions of nutrients to GBR waters may not necessarily translate into an increase in dissolved nutrient levels, local or regional increases in phytoplankton biomass would be anticipated. Such high phytoplankton biomass would also affect coral reefs either from increases in 'surplus' water column P concentration or the proliferation of benthic filter feeders resulting from indirect aquatic ecosystem changes.

Anthropogenic influences such as pumping treated sewage into nearshore waters or applying such effluent to golf courses can produce an increase in ambient levels of 1 order of magnitude or greater (Hopley, 1990). The fundamental impacts on coral reef communities from the disposal of nutrients derived from anthropogenic sources entails a noticeable deterioration of water quality, reductions in the diversity of corals by replacement with benthic flora and detrital fauna, and diminished aesthetics. More specifically, an elevated level of N promotes eutrophication, giving rise to widespread phytoplankton and attached algal blooms. The increased algal activity is in competition with the corals for space - destroying the reef structure by bio-erosion (Kinsey, 1988). Kinsey and Davies (1979) concluded that orthophosphate-P levels as low as 0.6 μM could cause a reduction in the calcification rate of coral by > 50% and thus retard the growth of the coral skeleton. Pastorok and Bilyard (1985) also found that increased P loads directly contributed to toxic decay of coral species. Moreover, sustained increases in water turbidity have occurred around ocean outfalls and non-point-source discharges, increasing the local coral mortality rates due to restricted light levels and increased sedimentation.

Bell (1992) suggested that effluent discharges rich in N transported via the terrestrial pathways of surface-runoff and groundwater seepage, or piped by ocean outfalls, could cause detrimental impacts on coral reefs. Thus, it is necessary to ensure an acceptable quality of sewage is discharged in the vicinity of coral reefs, whereby the use of effluent irrigation may retard the progress of N migration sufficiently to minimise this risk.

Various studies (Revelante and Gilmartin, 1982; Andrews, 1983 and Crossland and Barnes, 1983) have determined that in a broad sense, GBR lagoon waters have relatively high background levels, particularly in river-impacted regions. This poses a difficulty in achieving the required degree of dilution of sewage effluent in receiving waters.

Pastorok and Bilyard (1985) concluded that large outfalls in well-flushed open-coastal areas have minimal short-term impacts on coral reefs. Although, they noted that the effect of these discharges could be exacerbated when released to calmer inlets and bays within the GBR - whereby the prolonged residence time promoted algal growth and detritus formation.

Three reported examples of the impact of nutrient enrichment on fringing reefs adjoining resort island operations in the GBR lagoon deserve mention. Hopley (1982) described the effect of eutrophication resulting from a sewage outfall at Green Island, which promoted the growth of seagrass beds adjacent to the cay from about 900 m^2 in 1945 to over 130000 m^2 in 1978. Sand that eroded from the cay into the seagrass beds created a baffling effect that stabilised the sand on the reef flat and consequently led to its permanent displacement from the cay. Closer to the coastline, nutrient-rich effluent has been found to seep from septic saturated soils on Magnetic Island into local watercourses that ultimately flow onto the local fringing reef (Bell, 1990).

Monitoring undertaken at Hayman Island by van Woesik et al. (1992) found that the recruitment of corals was significantly diminished near the sewage outlet. Based on dye-tracer investigations undertaken at Hamilton Island, primary treated sewage effluent was expected to migrate north of the outfall during both ebb and flood tides (Water Quality Council, 1985a,b). This is likely to have contributed to the enhanced algal growth found north of the discharge point. The fringing reef of Catseye Bay at Hamilton Island has also been overgrown with algae. This may be a result of overland runoff from the resort which is rich in soluble-P (Bell et al., 1987).

1.2.2 Threshold Limits for Eutrophication

Coral reef ecology within the GBR is particularly sensitive to small increases in N and P levels above background as investigated by Connell and Hawker (1987), Bell et al. (1987) and others. They have reported eutrophication threshold levels of approximately 1 $\mu g\ L^{-1}$ for dissolved inorganic N (DIN) corresponding to between 2 and 3 times the ambient water levels. It should be noted that these studies were based on a 20% growth decrease of coral. Laws and Redalje (1979) ranked water quality parameters according to sensitivity toward eutrophication. Interestingly, they found that inorganic P was far more sensitive indicator than inorganic forms of N.

1.2.3 Mainland Riverine Loads into the Great Barrier Reef

In the total Great Barrier Reef Lagoon, major nutrient sources include mainland and island runoff and seepage, the adjacent ocean, and local N-fixing biota. The focus is on island runoff and seepage but mainland contributions need to be considered for comparative purposes when contemplating the anthropogenic induced changes from natural levels of nutrient inputs.

Mainland sources are difficult to quantify because of the huge variability in both stream flows and nutrient concentrations and the lack of data on groundwater discharges. The eventual fate of nutrient outputs from mainland sources is also unclear as substantial amounts, especially of P, may be trapped in coastal sediments. Brodie (1994) noted that for the whole GBR, sewage inputs are approximately one-tenth as large as diffuse runoff of nutrients resulting from agricultural activity.

The most comprehensive study of river outputs of nutrient to parts of the Great Barrier Reef Lagoon is that of Furnas et al (1994) who arbitrarily defined two areas or 'boxes' of the central GBR shelf. The more northerly one of these was designated the Cairns box, between Cape Tribulation and Cape Grafton, and the southern one the Tully box, between Cape Grafton and Dunk Island. The total shelf areas for the Cairns and Tully boxes were 5937 km^2 and 7826 km^2 respectively, while the estimates of volume of water were 197 km^3 and 312 km^3 respectively.

Furnas et al (1994) reported the periodic collection of dissolved and particulate nutrient samples from most of the major rivers which discharged directly into or immediately to the south of the two boxes. With the limited data available, a number of assumptions had to be made in order to produce an estimate of nutrient input to the respective boxes. Reasonably comprehensive discharge-weighted mean concentrations of N and P were available for only one major river, the South Johnstone River. Using these concentrations, estimates of the mass of nutrient exported from the other rivers were calculated using the average stream discharges determined hydrographically. The results suggested that the amounts of N transported annually on average to the Cairns and Tully boxes were approximately 2000 tonnes and approximately 4400 tonnes respectively. As the authors indicate, due to the number of assumptions involved, these estimates should be considered as provisional. Also, for reasons given in their report, the estimates are likely to be towards the upper end of the probable range. Corresponding estimates of P exports were 192 and 433 tonnes to the Cairns and Tully boxes respectively.

1.2.4 Overview of Great Barrier Reef Resort Island Water and Wastewater Management

Resort island water supply is obtained from various sources including surface storages, groundwater aquifers, and barging from the mainland. Wastewater treatment methods are most commonly either secondary or tertiary level and partial or full disposal can be achieved via land irrigation or ocean outfall, following detention in storage ponds.

A written survey and in many cases a field visit were conducted for several resort islands within the GBR Marine Park to ascertain their wastewater management practices. This information was later updated from the Great Barrier Reef Marine Park Authority (GBRMPA) database in 1999. Table 1 provides a summary of information relating to general physical characteristics, peak resort occupancy, level of sewage treatment and means of effluent disposal for a selection of GBR resort islands. Data relating to the quantity of wastewater produced, N and P concentration and total N (TN) and total P (TP) loads for a wide selection of resort islands are currently being compiled and will be detailed in a forthcoming effluent irrigation management report. However, relevant data for the islands examined in this report have been included in Section 2.1.4, 2.2.4 and 2.3.4.

Based on preliminary data, estimated TN and TP loads from individual resort islands are negligible in comparison to mainland nutrient contributions, and unlikely to exceed 1 tonne per year for TN and 0.1 tonnes per year for TP. Even if the TN load on each of the islands were exported to the marine environment, the consequence for the total nutrient budget of the Great Barrier Reef Lagoon would be negligible in comparison to the mainland riverine inputs. It should be remembered, however, that in addition to the uncertainties introduced by assumptions made in the calculation of the river inputs, little is known about the eventual fate of the nutrients once they reach the estuarine and coastal areas. It is possible that a substantial component of these nutrients is trapped in the near-shore areas. Detailed studies of sediment and water dynamics as well as measurements of water quality in these areas will be required to clarify the ultimate fate of the nutrients.

In spite of these unknowns it is clear that, if the nutrient contributions from resort islands is significant at all, it will be significant only in the marine environment in the immediate vicinity of the islands from which the nutrients emanate.

Table 1. General characteristics, peak occupancy and wastewater management for selected Great Barrier Reef resort islands

GBR Island Resort	Type	Area (km^2)	Peak Occupancy (Overnight / Day-trip*[1])	Sewage Treatment Level	Effluent Disposal Method
Brampton Island Resort	Continental	4.9	296 / -	Tertiary	Land irrigation at night on dry days and applied to golf course and gardens with a total area of 1.5 hectares
Daydream Island Resort	Continental	0.17	950 / -	Tertiary	Land irrigation covering north and south ends of the island on alternate nights from 0001 to 0700 hours, and ocean outfall for overflow
Dunk Island Resort	Continental	10	510 / -	Secondary	Land disposal area adjacent sewage treatment works
Great Keppel Island Resort	Continental	14	230*[2]/ 100*[2]	Secondary	Ocean outfall
Green Island Resort	Coral Cay	0.13	90 / 1900	Tertiary	10% land irrigation, 30% reticulated to amenities and 60% ocean outfall
Hamilton Island Resort	Continental	6	1600/ -	Secondary	70% land irrigation, 30% ocean outfall
Hayman Island Resort	Continental	4	750 / -	Secondary	Land irrigation
South Molle Island Resort	Continental	4	600 / -	Partial tertiary	Ocean outfall

*[1] Subject to availability of data

*[2] Average value

1.3 Great Barrier Reef Water Quality Guidelines

GBRMPA requires that all direct sewage discharges into the GBR Marine Park comply with tertiary treatment standards (Brodie, 1991). More explicitly, the nutrient quality of discharges must not exceed limits of total N and P concentrations of 4 mg L^{-1} and 1 mg L^{-1} respectively, as specified in GBRMPA (1993). However, secondary treated effluent may be discharged during wet weather at up to a maximum annual level of 5% of total annual outflow. Regulatory requirements governing proposals for land disposal of effluent are generally controlled by the pertinent local council, with tertiary treated sewage effluent being favorably considered, provided that adverse impacts on the quality of ground water resources are unlikely.

1.4 Nutrient Properties of Sewage Effluent

1.4.1 Nitrogen

Sewage effluent contains N in four different forms: organic N, ammonium (NH_4^+), nitrate (NO_3^-) and nitrite (NO_2^-). The treatment method affects the proportion of the different N components in the treated sewage effluent; for example, when sewage effluent treatment involves forced aeration, nitrification may take place and considerably increase the percentage of NO_3^- in the effluent.

Most of the N present in secondary treated effluent is in reduced forms, primarily NH_4^+ and organic species. Typically, 80% of the total N in effluent is NH_4^+ although values as high as 90 - 95% have also been recorded (Lance, 1972). Commonly, wastewater analysis entails a measure of the total Kjeldahl N (TKN) which is a measure of the combined NH_4^+ and organic N. The level of NO_2^- is generally negligible due to its rapid oxidation to NO_3^- in the presence of oxygen. Often, the concentration of NO_3^- in secondary municipal effluent is also relatively low.

Nitrate has a very low affinity for being sorbed onto soil surfaces. It is repelled by soil solids because they both possess a negative ionic charge. Due to the negligible adsorption property and high water solubility of NO_3^-, it is readily transported by infiltrating water through the vadose zone, and into ground water.

1.4.2 Phosphorus

Secondary sewage effluent often has a high P content, making it an important source of P for irrigated soils. The different forms of P include organic, condensed (pyro-, meta- and polyphosphates) and orthophosphates. The most prevalent chemically active P in soils is orthophosphate (Ryden and Pratt, 1980) as organic and condensed P decompose to orthophosphate in the soil or during treatment processes. The amount of P added to the soil through sewage effluent irrigation is then of particular interest given that, as indicated in Section 1.2.2, just a small increase in P above marine background levels can induce eutrophication.

There is generally minimal leaching of P through the soil due to the ability of most soils to sorb P onto mineral components such as clay and oxide surfaces (Haysom, 1974; Holford, 1989). The P sorption capacity has also been shown to be influenced by rainfall (Moody and Standley, 1979) and parent material (Standley and Moody, 1979).

An investigation of the P sorption capacity of various north Queensland soil groups by Moody and Chapman (1990) demonstrated that all but one soil type (siliceous sand) possessed moderate to high P buffer capacity. For siliceous sand, the leaching of applied P was restricted to 30 cm depth after 1700 mm rainfall (Teitzel, Standley and Abbott, 1983). Moody and Chapman (1990) thus concluded that a widespread loss of applied P by leaching would not occur from these soils. Additionally, Lance (1977) found that up to 90% of applied P in secondary sewage effluent was removed when added to calcareous sand for a period of 200 days consisting of 9 day flooding and 5 day redistribution cycles. Furthermore, adsorption of P continued even after the initial sorption capacity became saturated provided the infiltration rate did not exceed 150 mm d^{-1}.

Soil erosion is regarded as the principal transport mechanism for P, which often becomes attached to soil particles during overland runoff processes. Soluble P associated with sediment in surface runoff was been found to vary linearly with the application rate of P (Romkens and Nelson, 1974).

This study therefore concentrated on N because, in land disposal of sewage effluent, P is much more likely to bind to the soil and consequently its transport through soils is more difficult to monitor given the scope of this project.

1.5 Nitrogen Cycle

As soon as sewage effluent reaches the soil, it becomes part of the soil N cycle. Since the N content of soil organic matter is reasonably constant at around 5%, observed changes in soil N content often reflect changes in the inorganic (mineral) species of N. Mineral forms of N that occur in soils are the plant-available forms of NH_4^+ and NO_3^-. After NO_3^- and NH_4^+ are taken up by grass, they eventually become immobilised as soil organic matter when grass residues are returned to the soil. Immobilisation is a biological process influenced in part by the soil temperature, soil pH and the chemical form of the available inorganic N while mineralisation is the converse transformation of organic N into an inorganic form. The most important factor in determining whether there is net mineralisation or immobilisation is the C:N ratio of the organic material. High C:N ratios lead to net immobilisation of soil N and low C:N ratios lead to net mineralisation of soil N. Both of these processes occur simultaneously in soils, hence the net result dictates whether N will be available for plant use.

The NH_4^+ contained in the effluent, as well as that derived from organic N, is usually oxidized to NO_3^- by nitrification. In soil-water systems, nearly all NO_3^- reactions are microbiological. Losses of NO_3^- by leaching below the rooting depth and as gas (N_2 or N_2O) by denitrification are often large. As soil conditions change, the bacteria controlling NO_3^- reactions may also change. For example, the presence or absence of oxygen in the soil plays a major role in which bacterial populations are active, and subsequently which NO_3^- reactions occur. Denitrification is thus affected by many species of microorganisms, which utilize NO_3^- as a terminal electron acceptor in waterlogged and anaerobic environments. The flow of N in turfgrass soils is summarized in Figure 1.

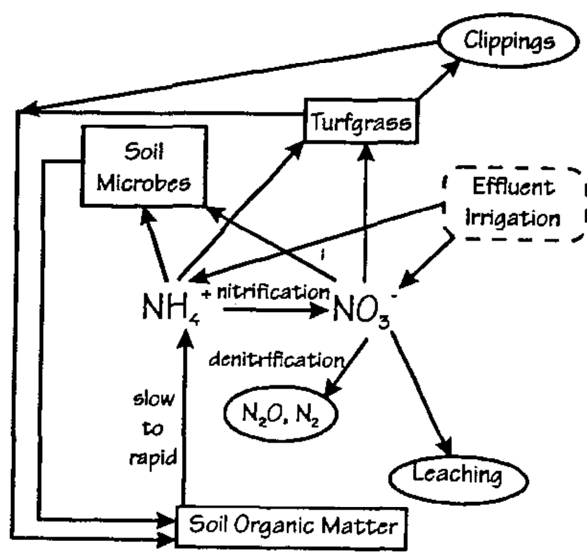

Figure 1. The major flows of N in turfgrass soils.

1.6 Studies of Nitrate Leaching Under Turfgrass: A Review

Numerous methods of measuring N migration have been adopted in studies involving the leaching of N applied to turfgrass. These entail the sampling of soil water in the vadose zone, the collection of leachate from drainage, soil sampling and observation piezometers in groundwater systems. Provided there was only minimal upward movement of water from below the root zone, it was generally assumed that once NO_3^- leached beyond the root system it would eventually pass freely into the underlying aquifer. A review of previous studies involving N leaching below turfgrass showed that outcomes were highly variable and influenced by soil type, irrigation, N source, N application rates, and season of application.

A study by Brown et al. (1977) examined the leaching of N from combined Tifdwarf bermudagrass and perennial rye grass plots in a variety of sand (80 - 85%), clay (5 - 10%) and peat (10%) mixtures. Several fertilisers were applied sporadically and irrigated at rates of 0.6 - 0.8, 0.8 - 1.0 and 1.0 - 1.2 cm per application. As the application rate increased from 24 to 98 kg ha^{-1} of applied N, the NO_3^- lost in leachate decreased from 37.8 to 15.5%. Associated with this decrease in leaching fraction however, was a rise in the total loss of N from 9 to 15 kg ha^{-1}which had direct ramifications on the concentration of N in drainage. Additionally, they found that when the irrigation rate was maintained at or below the evapotranspiration (ET) rate, the degree of NO_3^- leaching from soluble inorganic N sources was minimised. Plots experiencing low application rates of irrigation did not demonstrate any peak concentrations of NO_3^- during their study.

However, when a fine sandy loam soil was used as the rooting zone media, the proportion of fertilizer N that leached as NO_3^- was reduced from 14.6 to 4.6% as the N rate increased. More importantly, the amount of NO_3^- was essentially unchanged as the N rate increased. It is likely that increased grass growth and hence nutrient uptake, was associated with increasing N rates, thus decreasing leachate volume.

Morton et al. (1988) investigated the leaching losses of NO_3^- from plots of Kentucky bluegrass turf subjected to three levels of fertilization (0, 97 and 244 kg N ha^{-1} yr^{-1}). Two different irrigation regimes were scheduled to either (i) avoid drought stress in the turfgrass or (ii) simulate over-watering. Over a two-year period, mean annual losses ranged from 2 kg N ha^{-1} on the unfertilized, minimum irrigation control plot to 32 kg N $ha^{-1}$for the over-watered, high N application.

Seasonal decreases in turfgrass uptake at the time of fertiliser application could have exacerbated leaching losses due to excess soluble N in the root zone. Furthermore, increases in N drainage from both the control and low treatment plots were associated with significant rainfall events.

A study was conducted by Mancino and Troll (1990) to determine NO_3^- leaching from fertilised and irrigated 'Penncross' creeping bentgrass turf growing in 80% sand and 20% peat rooting media. For ten weeks, a range of fertilisers were applied at 19.5 kg N ha^{-1} in conjunction with irrigation equivalent to 38 mm per week. While 46% of the irrigated water was leached, total leaching losses of N were < 0.5% of the applied N. In comparison to a single application of 49 kg N ha^{-1}, marginal increases were produced with total N leached < 4.1% for quick-release fertilisers and negligible for slow-release types. Consequently, they concluded that N leaching losses from fertiliser applied to turf on sandy soil mixtures could be low even when irrigated at moderately heavy rates.

Soil texture can be an important factor in the leaching of N from turfgrass plots due to its influence on the rate of denitrification, total amount of leachate and to the ability of soil to retain NH_4^+. Rieke and Ellis (1974) undertook a 2-year study of N leaching from turfgrass plots on sandy soil which were fertilised with three equal applications of 390 kg N ha^{-1} ammonium-nitrate. As anticipated, soil NO_3^- concentrations were pronounced in the top 30 cm of the soil during a majority of the study period. However, when compared to control plots, notable increases in NO_3^- concentration at a soil depth of 45 to 60 cm were measured in only two of the 20 samplings. This outcome suggests only a limited potential for NO_3^- leaching for sites experiencing similar conditions.

Rieke and Ellis (1974) also studied the N leaching response of sandy loam soil using identical procedures to the study they conducted on sand. None of the treatments increased soil NO_3^- concentrations in the 45 to 60 cm soil depth over concentrations measured in the control plots. As for the sandy media, soil NO_3^- concentrations at the surface were more elevated but deeper migration of NO_3^- was not encountered.

Synder et al.'s (1981) 2-year study of N losses from various fertilisers applied at 80 kg ha^{-1} bimonthly to bermudagrass sand greens showed that the average NO_3^- leaching loss for urea was only 1% of the applied N. The corresponding mean NO_3^- concentration in the collected leachate was approximately 0.2 mg L^{-1}which is far below the safe drinking water standard. For all other fertilisers used, no greater than 9.3% of applied N was lost to drainage ($CaNO_3$).

At a low rate of 39 kg N ha^{-1}, applied bimonthly, they noted very little leaching of any N source. The highest leaching of inorganic N was 2.9% of applied N for the duration of the study.

Brown et al. (1982) measured the effects of different sources of N in leachate emanating from bermudagrass golf greens overlaying a variety of rooting mixtures ranging from sand to sandy-loam. Irrigation rates were designed to exceed ET and generate adequate leachate volume, with 1 cm d^{-1} applied between May and September and 1 cm every other day during the remainder of the year. A single application of 163 kg N ha^{-1} was adopted for ammonium-nitrate and 146 kg N ha^{-1} for all other fertiliser types except urea at 244 kg N ha^{-1}. The greatest NO_3^- loss occurred from ammonium-nitrate applications and ranged from 8.6% (sandy-loam) to 21.9% (sand). All NO_3^- losses from other N sources ranged between 0.1 and 9.5% applied N. This suggested that fertilisation with organic forms of N was a preferred means of limiting N losses and that sandy-loam was more effective in utilising applied N than sands.

Literature on NO_3^- leaching from turfgrasses grown on finer-textured soil is more scarce with long-term field data lacking. Nelson et al. (1980) investigated the losses of urea applied at 253 kg N ha^{-1} to Kentucky bluegrass growing on either thatch or 5 cm of a silt loam soil. After 15 days they found that 32 and 81% of the applied urea leached as NO_3^- from the silt loam soil and thatch, respectively. They believed that NO_3^- leaching was enhanced during the passage of moisture through the profile if NO_3^- was soluble and in concentrations that exceeded that utilised by turf. If N was not readily available in the soil, leaching losses would be diminished.

Most of the studies were conducted under the "worst case scenario," whereby soil-grass systems were heavily irrigated and fertilized at several times the normal use rate. These approaches were often contrasted with less extreme conditions or experimental controls.

These results clearly illustrate that NO_3^- can be lost from turfgrasses grown on sandy soils whereas on sandy loam greens, increased N fertilization may not compromise the leachate quality. The use of slow-release fertilizers has also been shown to reduce or eliminate leaching losses in several studies (Rieke & Ellis, 1974; Nelson et al., 1980; Brown et al., 1982). Unfortunately, most of these studies have only monitored losses over a 2 to 3 year period. Longer term studies may determine if a turfgrass system has a maximum N load that it can handle or retain, beyond which N would be lost through leaching. It may be expected that this maximum N load would be reached sooner where clippings are returned.

2. FIELD SITES

Modelling work has to date concentrated on three islands of varying hydrogeology and markedly differing climatic conditions. Currently, Great Keppel Island resort discharges effluent to the sea via a piped outfall, but plan to dispose treated effluent on lawns, golf courses and gardens by irrigation in the future. Dunk Island discharges sewage effluent to an adjacent land disposal area. Brampton Island currently employs a scheme of effluent irrigation and does not have a piped outfall for effluent.

2.1 Great Keppel Island

2.1.1 Location and Climate

Great Keppel Island is a high continental island located in the GBR region and is situated 50 km northeast of Rockhampton at 23°10' S latitude and 150°58' E longitude. It has a total area of approximately 1400 ha, the primary land-use being a resort and camping ground. In general, the Great Keppel Island area experiences a sub-tropical, sub-humid climate with hot wet summers and mild dry winters.

2.1.2 Geology

According to Lloyd (1980a), the geology of the resort area consists of three formations. Two Quaternary formations are underlain by Paleozoic bedrock. The two Quaternary formations are (i) Holocene/Pleistocene Dune Sands and (ii) Holocene Outer Barrier Deposits.

The former consists of white, uniform, fine-grained sands overlying a bed of shell grit, while the outer barrier deposits consist of darker fine-grained quartz sands overlying a similar shell grit material. Lloyd (1980a) suggested that both deposits should be considered as one unit for the purposes of groundwater examination, due to the similarity of the sieve analyses and aquifer permeability.

2.1.3 Water Supply and Irrigation

Throughout the resort lease, there are a number of groundwater bores that are utilized for domestic and irrigation purposes (Figure 2).

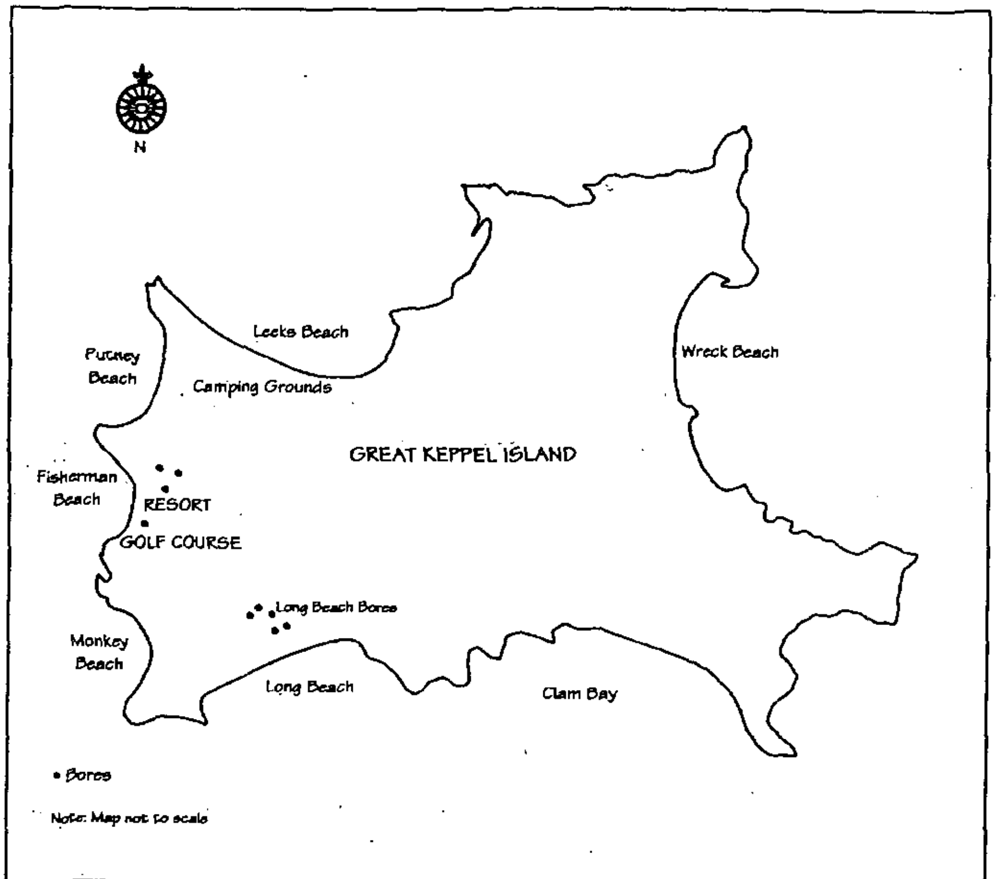

Figure 2. Land use and production bore locations (dots) on Great Keppel Island.

Values of domestic water consumption and irrigation rate of groundwater are presented in Table 2.

Table 2. Annual consumption of groundwater on Great Keppel Island for domestic supply and irrigation.

Water Source	Water Usage (ML)						
	1987	1988	1989	1990	1991	1994	1995(1)
Resort bores (domestic)	86.8	38.6	96.4	85.5	35.4	20.3	9.5
Long Beach bores (domestic)	18.2	82.8	33.7	47.2	80.5	93.2	14.7
Total domestic consumption	*105.0*	*121.4*	*130.1*	*132.7*	*115.9*	*113.5*	*24.2*
Golf Course bore (irrigation)	5.4	6.8	19.3	23.9	35.0	31.5	6.1
Gardens bore (irrigation)	5	12.5	19.0	31.8	39.0	24	4.9
Total irrigation	*10.4*	*19.3*	*38.3*	*55.7*	*74.0*	*56.3*	*11.0*

Note (1) June-September period.

Lloyd's (1980a) bore investigations within the present golf-course area indicate a relatively shallow water table in this area. Two bores drilled just over 1 m apart behind the private dwellings adjacent the golf course, were shown to have water table depths approximately 2 m below the surface.

Lloyd's (1980a) report also gave details of aquifer pumping tests conducted on each of the production bores. Water levels were monitored in adjoining observation bores enabling an estimate to be obtained of transmissivity, which is the ability of the aquifer to transmit water through a unit width for a unit hydraulic gradient. The average value from these tests was found to be 110 m^2d^{-1}.

The water quality of these bores has been regularly checked for suitability as drinking water. At present however, this analysis does not include any species of N. The water quality is generally very good, although there is some evidence of seawater intrusion in older spears in vicinity of Long Beach.

2.1.4. Effluent Quantity and Quality

Wastewater is treated to secondary stage on Great Keppel Island. The processes include screening, flow balancing, extended aeration, clarification and chlorine disinfection. At the time of provision of the data, effluent was being discharged to the ocean via an outfall with diffuser.

Effluent flow rates have been monitored by a flow meter in the inlet pipe to the secondary treatment works. Statistics for wastewater flows are given in Table 3.

Table 3. Great Keppel Island effluent flow data.

Effluent Quantity	**Monitoring Period**	
	January – December 1994	**June - September 1995**
Resort effluent volume	59 ML	17.7 ML
Average day flow	164 kL d^{-1}	145 kL d^{-1}
Sewage flow / domestic water usage	0.51	0.73

The nutrient concentrations of the effluent water (NO_3^-, TKN and TN) are shown in Figure 3 below. These data were obtained from analyses commissioned by the resort and undertaken by water and environmental analysts and consultants, Simmonds & Bristow Pty. Ltd.

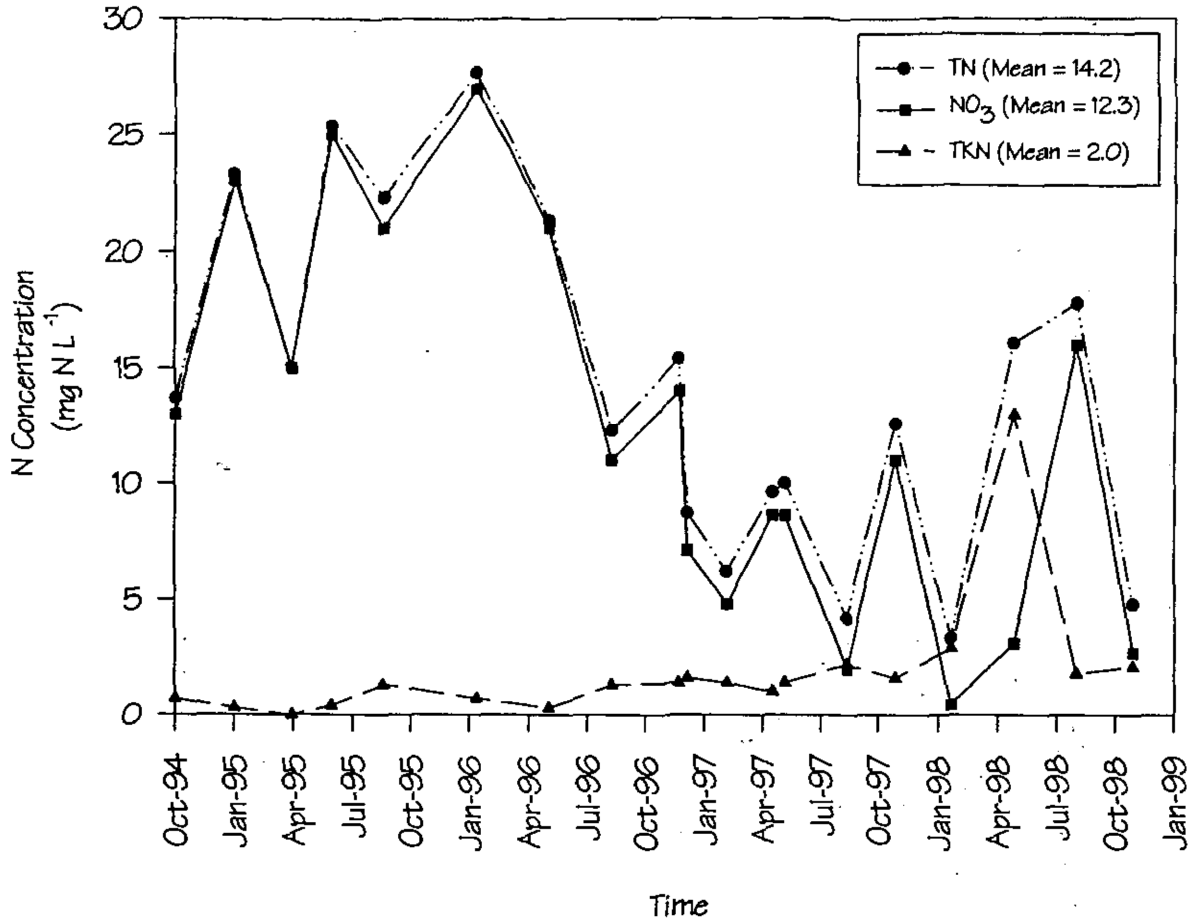

Figure 3. Concentration of various forms of N (TN= total N, NO_3 = nitrate, TKN = total Kjeldahl N) in final effluent from Great Keppel Island resort.

2.2 Dunk Island

2.2.1 Location and Climate

Dunk Island is a continental island in the northern GBR region, located at 17°57' S latitude and 146°10' E longitude. This 1000 ha island is characterized by tropical conditions and high annual rainfall.

2.2.2 Geology

Based on the drilling logs of three of the production bores within the resort lease, red loamy clay extends down from the natural surface level until the water table is reached at approximately 20 m below natural surface level, revealing an aquifer of fractured Hodgkinson shale and slate. Previous geological studies of the island have shown that the southeastern half of the island is composed primarily of younger granitic rocks, which prominently feature quartz diorite of the Tully granite complex.

2.2.3 Water Supply and Irrigation

Potable water for the resort is provided by four groundwater production bores within the resort lease (see Figure 4). Following treatment, this is reticulated throughout the resort for all internal uses, including washing and ablutions. Golf greens and gardens are currently irrigated with dam water, but fairways are not currently irrigated. There are also a number of small perennial streams on the island.

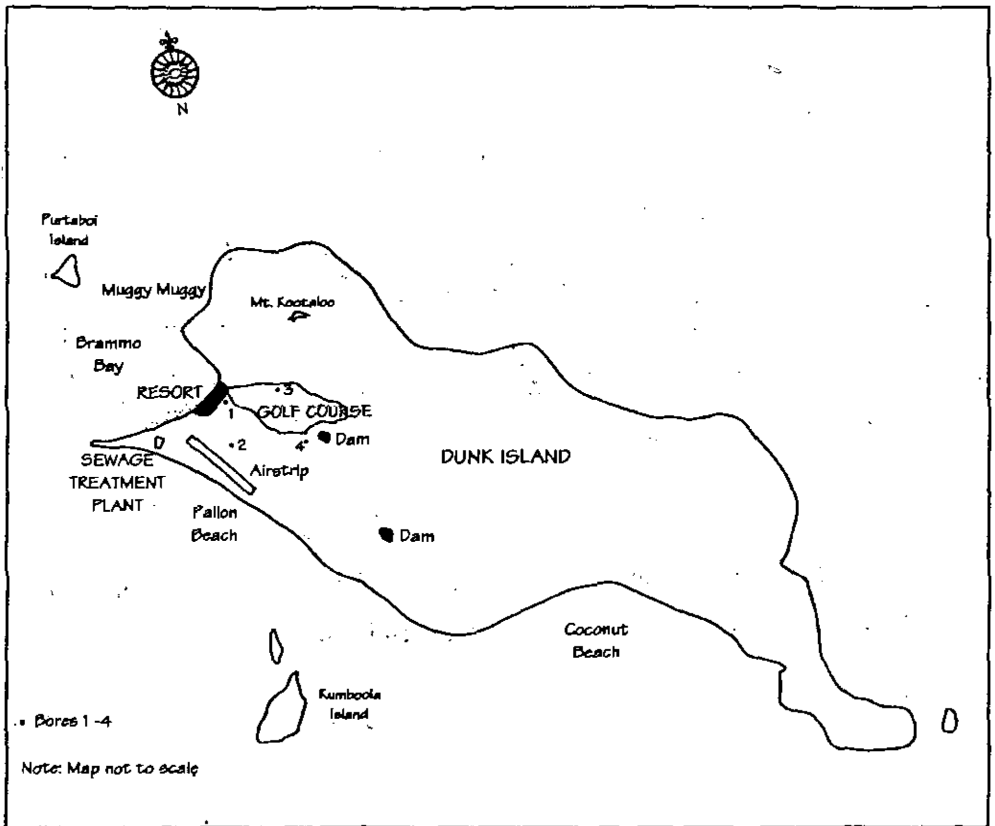

Figure 4. Land use and production bore locations (1-4) on Dunk Island.

2.2.4 Effluent Quantity and Quality

The sewage treatment plant is currently configured to provide primary screening with a fine rundown screen followed by a sequential decant activated sludge system for treatment of the raw sewage. Effluent is discharged to an adjacent land disposal area and sludges are disposed to the old decant clarifier tank at the treatment plant. Collected sludge is allowed to degrade anaerobically and is then pumped onto sludge drying beds for dewatering.

Effluent flow rates are currently only measured as inflow to the plant. Based on data obtained from the resort for 1995, average daily flow was approximately 200 kL d^{-1}. Measured values for NO_3^-, TKN and TN are shown in Figure 5 below. The water analyses were commissioned by the resort and undertaken by water and environmental analysts and consultants, Simmonds & Bristow Pty. Ltd.

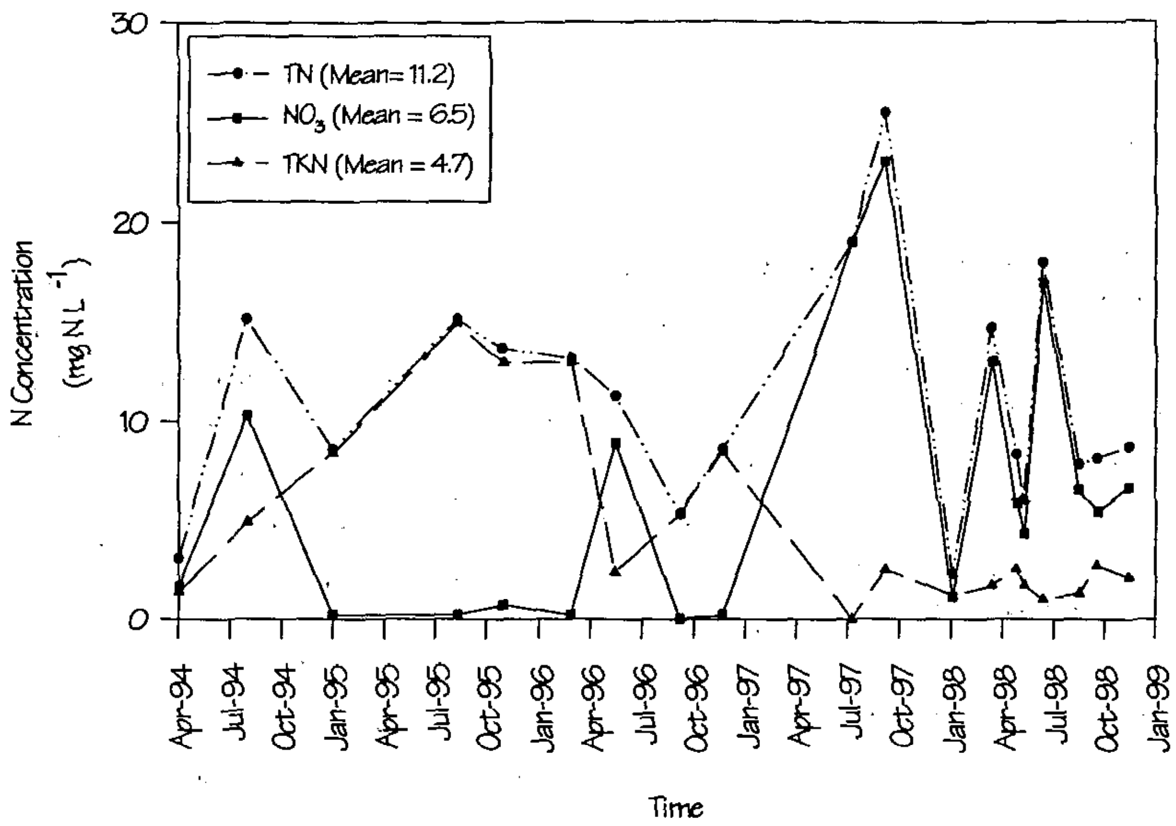

Figure 5. Concentration of various forms of N (TN= total N, NO_3 = nitrate, TKN = total Kjeldahl N) in final effluent from Dunk Island resort.

2.3 Brampton Island

2.3.1 Location and Climate

Brampton Island forms part of the Cumberland Islands Group, situated at latitude 20°49' S and longitude 149°17' E. The primary land use is designated National Park, although a small resort operation is located on the island which encompasses approximately 20 ha of the island's 485 ha total area. The area experiences a sub-tropical, sub-humid climate. Daily rainfall data from the Bureau of Meteorology were available for this site, while daily values of temperature and relative humidity were taken for nearby Mackay.

2.3.2 Geology

The majority of the island consists of andesitic crystal tuffs and flows that overlie granites. In outcrop, these volcanics are generally fresh with little weathered rock and range from close jointed spacing (25 - 50 mm) to massive joints over 1 m apart (Lloyd, 1980b). Dune sands are associated with Dinghy Bay and Western Bay (see Figure 6), although these are not extensive and are not highly elevated above sea level. Particle size distribution tests on various soil samples taken at 1 m below surface level have shown the material to be of a sandy-clay type: brown, fine to coarse sand with some rock fragments.

2.3.3 Water Supply and Irrigation

Three dams are used for water supply, located at Western Bay West (WBW), Western Bay East (WBE) and adjacent the airstrip. However, there are no perennial rivers or streams on the island. A small network of groundwater wells is also used for domestic and irrigation purposes. The bore logs of two bores (Petts No. 1 and No.2) located within the resort show water-bearing rock occurring near to granite contacts. The sole water bed recorded in Petts No.1 was at 35 m below natural surface level, while in Petts No. 2 water beds are recorded above and below the granite reported at 26-30 m and again immediately above the fresh granite reported at 46-48 m below natural surface level.

According to groundwater drilling investigations commissioned by the resort in 1986/87, the total reliable yield calculated over the dry season of July to December was estimated to be 8 ML. This was based on information obtained from pump-tests of thirteen bores installed in the vicinity of Western Bay and Dinghy Bay. Water samples were analysed for compliance with drinking water standard and general quality. Several bores contained unacceptably high levels of dissolved salts. The remaining bores were deemed to provide acceptable water quality for general use.

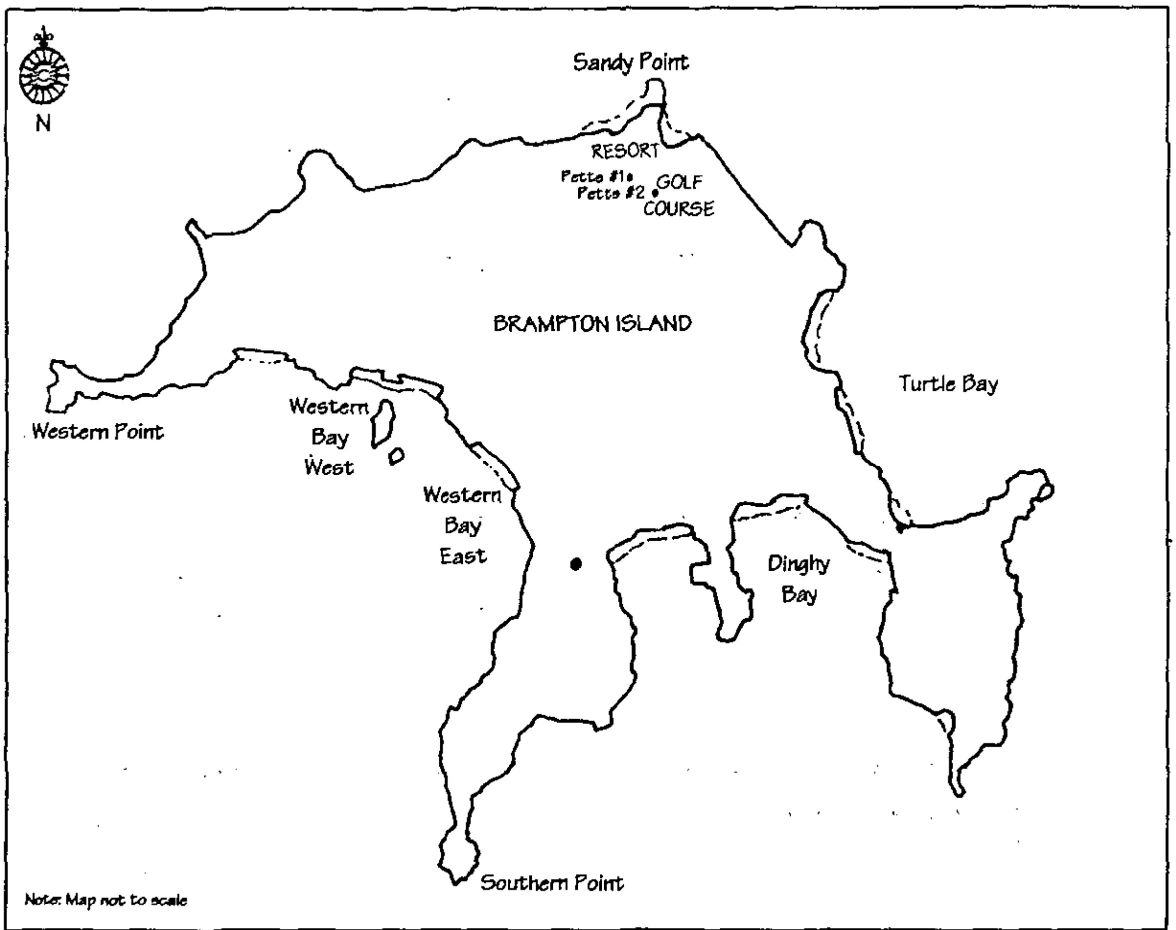

Figure 6. Land use and resort bore locations on Brampton Island.

2.3.4 Effluent Water Quantity and Quality

Secondary wastewater treatment is carried out on the island. The quantity of effluent produced each day, equates on average 55 kL. The majority of the treated effluent is irrigated over the golf course during the night. Treated effluent is also used on some of the lawn area and garden beds. On most occasions during dry periods all treated effluent is used. The irrigation of the golf course is achieved using a water distribution network of 24 sub-surface stations connected to an array of pop-up sprinklers. The irrigated area is estimated to cover no more than 1.5 hectares.

Effluent quality analysis was regularly undertaken by water and environmental analysts and consultants, Simmonds & Bristow Pty. Ltd., the results of which are shown in Figure 7. Solid waste from the treatment process is transported back to the mainland.

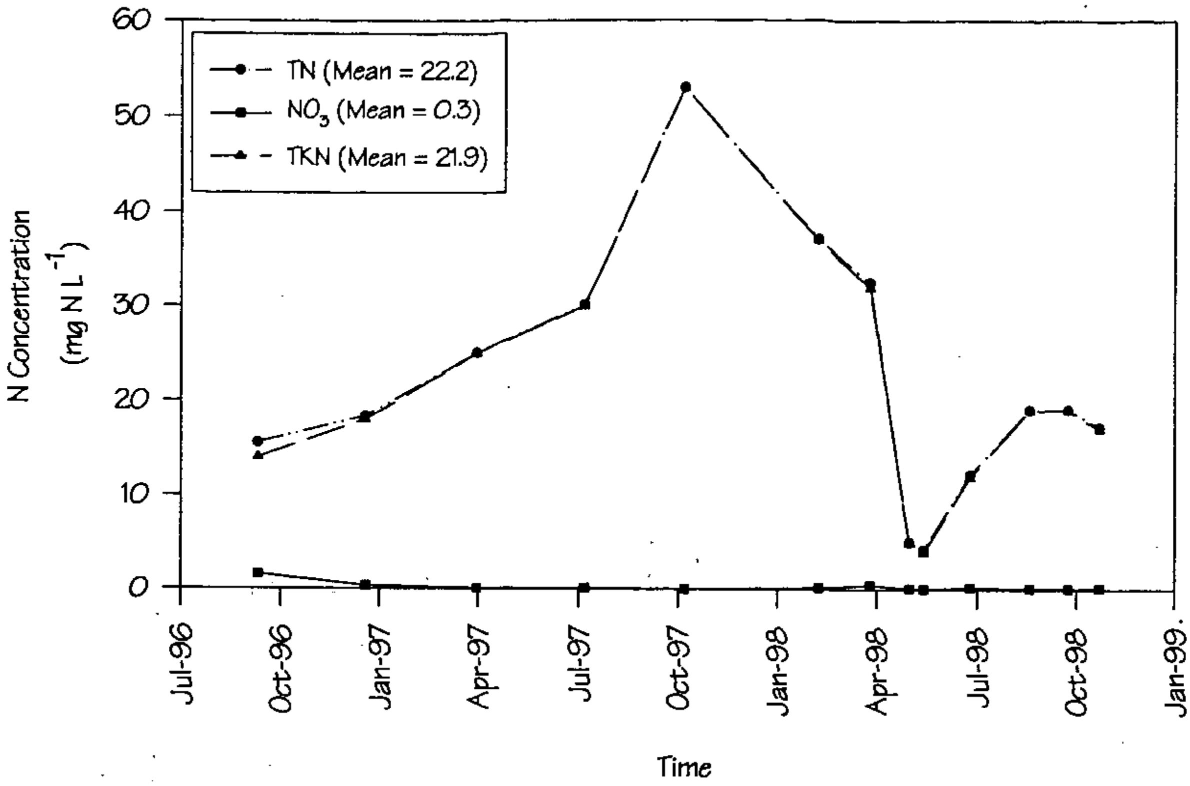

Figure 7. Concentration of various forms of N (TN= total N, NO_3 = nitrate, TKN = total Kjeldahl N) in final effluent from Brampton Island resort.

3. NUMERICAL MODELLING OF WATER AND NITROGEN FLOW BELOW THE ROOT ZONE

Two inter-linked models were used to simulate soil-water transport and N dynamics within the root zone resulting from effluent irrigation applied to turfgrass. The models: SOIL (Jansson, 1991) and SOILN (Eckersten et al., 1996) were developed at the Swedish University of Agricultural Sciences in Uppsala. The fundamental principles of each model are outlined below.

3.1 The SOIL model

SOIL adopts a physically based approach to simulating the movement and storage of water within a one dimensional soil-plant-atmosphere continuum. The driving variables of the model consist of standard daily meteorological data i.e., air temperature, relative humidity, rainfall, irrigation, global radiation and wind velocity. The major water flows accounted for in the model are depicted in Figure 8.

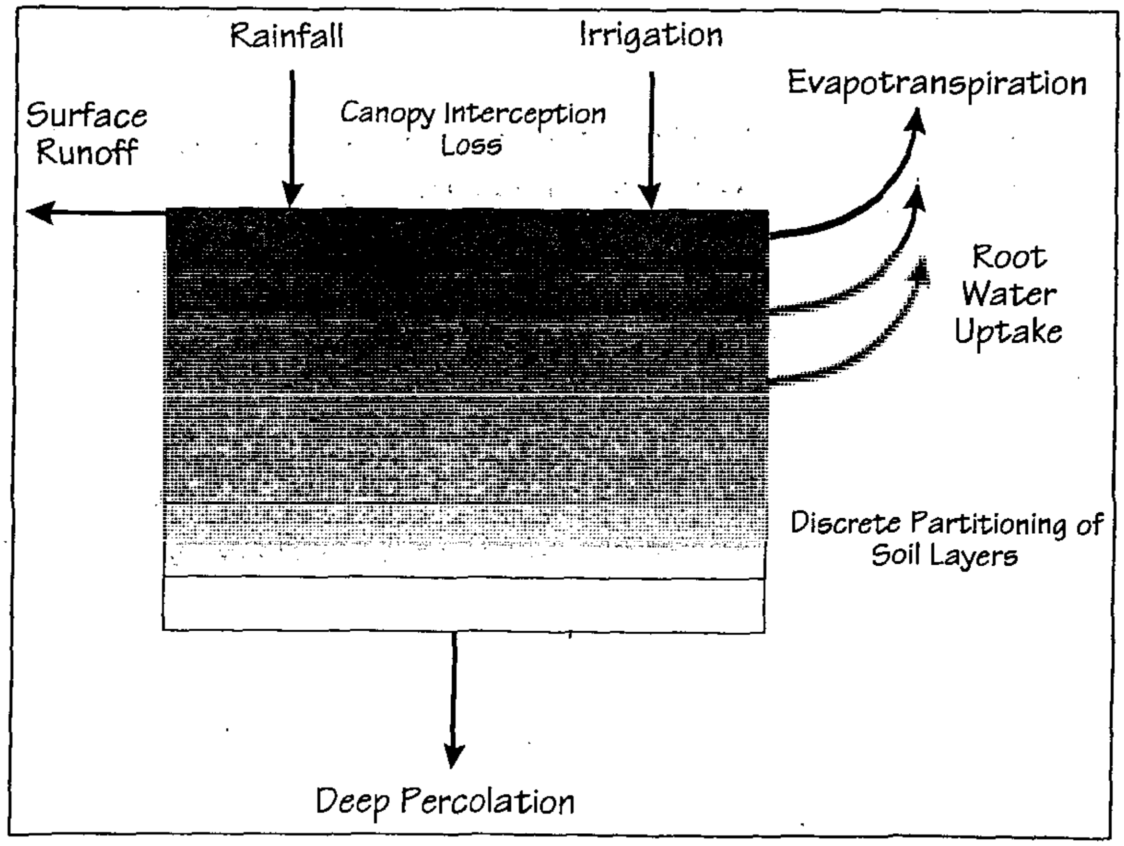

Figure 8. Soil water flows in the soil-plant-atmosphere system as represented by SOIL.

3.1.1 Fundamental Processes and Equations

Unsaturated soil water flow, q_w, is based on the partial differential equation derived from Darcy's law for laminar flow:

$$q_w = -K\left(\frac{\partial \psi}{\partial z} - 1\right) \qquad (1)$$

where ψ is the suction of water, K is the unsaturated hydraulic conductivity, z is the depth from the soil surface.

This is substituted into the continuity equation:

$$\frac{\partial \theta}{\partial t} = -\frac{\partial q_w}{\partial z} + s_w \qquad (2)$$

to give Richards' equation (Richards, 1931):

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left(K\frac{\partial \psi}{\partial z} - K\right) + s_w \qquad (3)$$

where s_w is a source/sink term representing the net outflow of water caused by root water uptake or drainage to groundwater and θ is the volumetric water content.

The soil profile is divided into discrete layers that are treated separately regarding storage and flows. Two soil hydraulic functions must be known to solve equation (3), the water retention curve and the hydraulic conductivity function.

Experimental water retention data in the intermediate range of suctions are described by the water retention curve in the functional form given by Brooks and Corey (1964):

$$S_e = \left(\frac{\psi}{\psi_a}\right)^{-\lambda} \qquad (4)$$

where ψ_a is the air-entry suction and λ is the pore size distribution index.

Effective saturation, S_e is defined as:

$$S_e = \frac{\theta - \theta_r}{\theta_s - \theta_r} \tag{5}$$

where θ_s is the porosity and θ_r is the residual water content. Calculation of the parameters λ, ψ_a and θ_r is done by least squares regression of equations (4) and (5) to experimental water retention data. In order to achieve a representative fit over the whole range, equations (4) and (5) are fitted only to data corresponding to suctions below a threshold value, ψ_x. This represents the point of transition to log-linear behaviour in $\theta(\psi)$ as observed in measured data.

Thus, the relation between water content and suction above this threshold is assumed log-linear:

$$\frac{\log\left(\frac{\psi}{\psi_x}\right)}{\log\left(\frac{\psi_{wilt}}{\psi_x}\right)} = \frac{\theta_x - \theta}{\theta_x - \theta_{wilt}} \qquad \psi_x < \psi < \psi_{wilt} \tag{6}$$

where θ_x ($=\theta(\psi_x)$) is the threshold water content and θ_{wilt} is the wilting point, defined as the moisture content at a suction of 15 bar water.

For moisture contents approaching saturation, i.e. from θ_s to θ_m, a linear expression is used for the θ - ψ relationship.

$$\psi = \psi_m - \frac{(\theta - \theta_s + \theta_m)}{\theta_m}\psi_m \tag{7}$$

where θ_m is the estimated macropore volume (vol %) and ψ_m is the suction which corresponds to a water content of θ_s - θ_m.

Subsequently, the unsaturated conductivity K is calculated using the analytical expressions according to Brooks & Corey (4) and (5) and the expression given by Mualem (1976):

$$K = K_{sat} S_e^{\left(n+2+\frac{2}{\lambda}\right)} \tag{8}$$

where K_{sat} is the saturated conductivity and n is a parameter accounting for pore correlation and flow path tortuosity.

Total evaporation from the system, E, is divided into three main components, evaporation from soil (E_S), transpiration (E_{Ta}) and evaporation of water intercepted by the canopy (E_{Ia}).

$$E = E_S + E_{Ta} + E_{Ia} \qquad (9)$$

Each component is calculated from Penman's combination equation in the form given by Montieth (1965). Actual transpiration is calculated from potential transpiration accounting for the depth distribution of roots and soil water suction. Thus, water is taken up in proportion to the root fractions in each layer until a critical suction is reached, whereupon turfgrass water uptake is reduced.

Boundary conditions at the soil surface are dictated by the infiltration capacity there as calculated from the saturated conductivity of the topsoil assuming a unit gradient. If the infiltration capacity of the uppermost soil layer is exceeded, water ponds on the soil surface. This moisture can either infiltrate into the soil with a delay or be lost as surface runoff. The surface runoff, q_{surf}, is calculated as a first order rate process:

$$q_{surf} = a_{surf}(W_{pool} - w_{p\max}) \qquad (10)$$

where a_{surf} is an empirical coefficient, W_{pool} is the total amount of water in the surface pool and w_{pmax} is the maximum amount which can be stored on the soil surface without causing any surface runoff.

A more detailed description of the SOIL model is found in Jansson (1991).

3.1.2 Selected Input Parameters

The primary focus of this numerical study was to ascertain the long-term consequences (for different island hydrogeologies, environmental conditions and wastewater management) of effluent irrigation applied daily to a turfgrass area. The simulation period was performed for a daily time-step over twenty consecutive years from 1st January 1974 to December 31st 1993.

3.1.2.1 Hydraulic Loads

Although Brampton Island currently irrigates their effluent, only limited irrigation flow data are available. Hence, for the purpose of this study a series of hypothetical irrigation loading rates based on fractions of the total daily effluent production from its sewage treatment plant were applied to the golf course area on a daily basis. The selected rates were 0, 25, 50, 75 and 100% and were similarly applied to Great Keppel and Dunk Islands (refer to Table 4).

Table 4. Proposed effluent irrigation hydraulic loading rates for SOIL modelling

Fraction of total daily effluent irrigated [%]	**Great Keppel Island (Area = 1.2 ha)**	**Dunk Island (Area = 5 ha)**	**Brampton Island (Area = 1.5 ha)**
	Hydraulic loading [mm d^{-1}]	**Hydraulic loading** [mm d^{-1}]	**Hydraulic loading** [mm d^{-1}]
0	0	0	0
25	3.3	1.0	0.9
50	6.7	2.0	1.8
75	10.0	3.0	2.8
100	13.3	4.0	3.7

3.1.2.2 Climatic Conditions

Data sets of daily air temperature, relative humidity, rainfall, wind speed, global radiation and irrigation served as the driving variables (time-dependant physical inputs) for the SOIL model.

For each site, wind speed was assumed constant at 3 ms^{-1}. Site-specific daily rainfall records for each of the islands were obtained from the Bureau of Meteorology for the period of investigation (shown cumulatively in Figure 9), although air temperature, relative humidity and global radiation data were not available. Thus, daily air temperature and relative humidity data for Dunk, Great Keppel and Brampton Islands were obtained for geographically similar locations – Cardwell (18.26 S, 146.02 E), Yeppoon (23.1 S, 150.73 E) and Mackay (21.16 S, 149.12 E) weather stations respectively. Daily global radiation values for Great Keppel and Brampton Islands were based on data recorded at Rockhampton AMO (23.38 S, 150.47 E), while Townsville AMO (19.25 S, 146.76 E) was chosen as the most representative site for Dunk Island. For illustrative purposes this data is shown as mean monthly values (Figure 10, 11, 12).

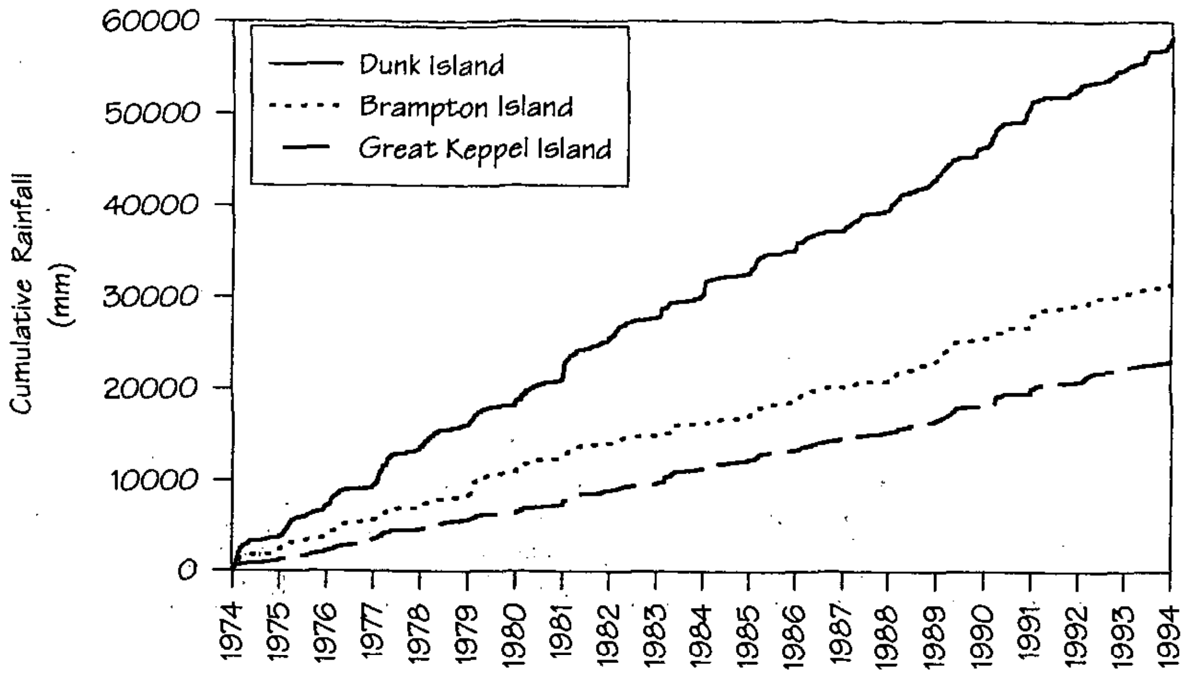

Figure 9. Cumulative rainfall for selected GBR resort islands during 20-year study.

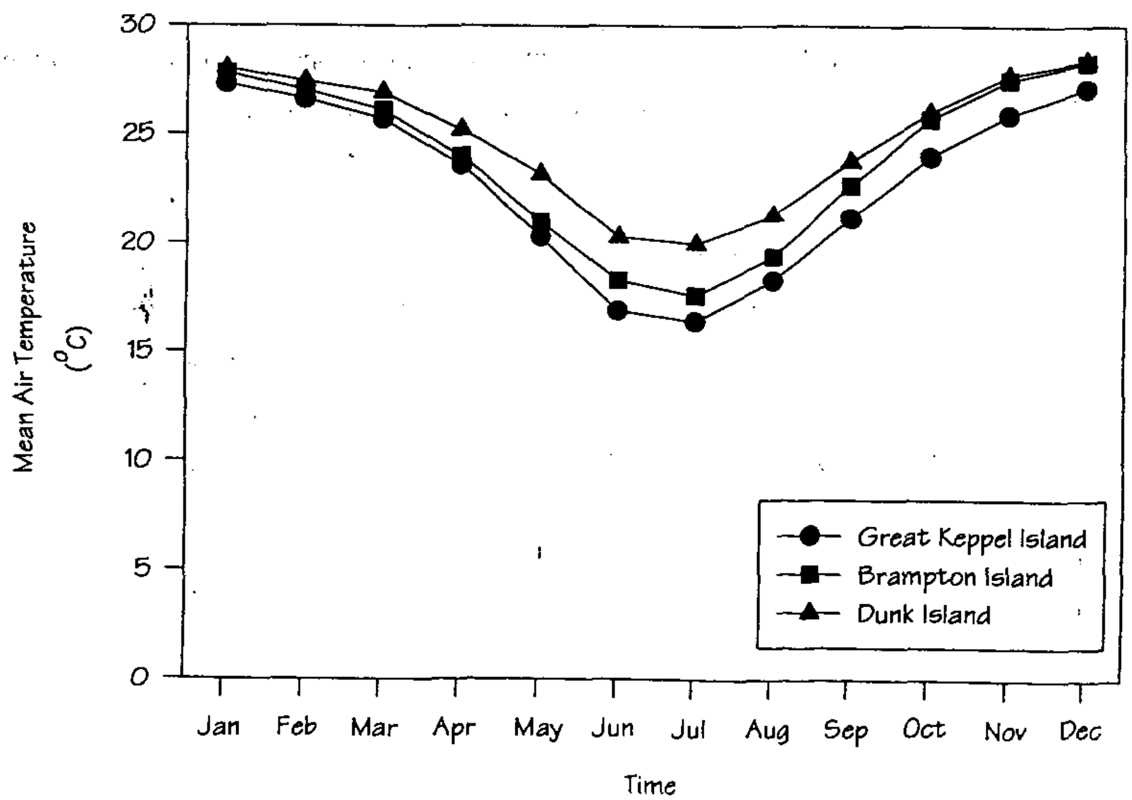

Figure 10. Mean monthly air temperature at 9am adopted for selected GBR resort islands.

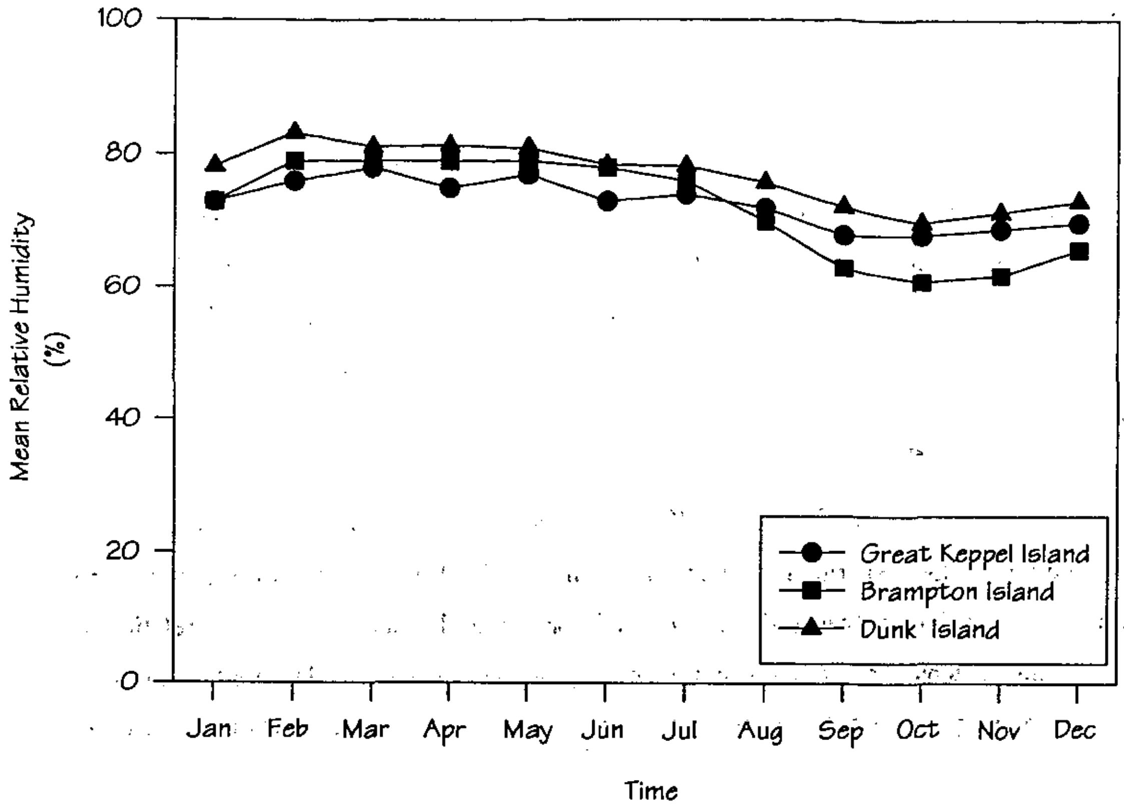

Figure 11. Mean monthly relative humidity at 9am adopted for selected GBR resort islands.

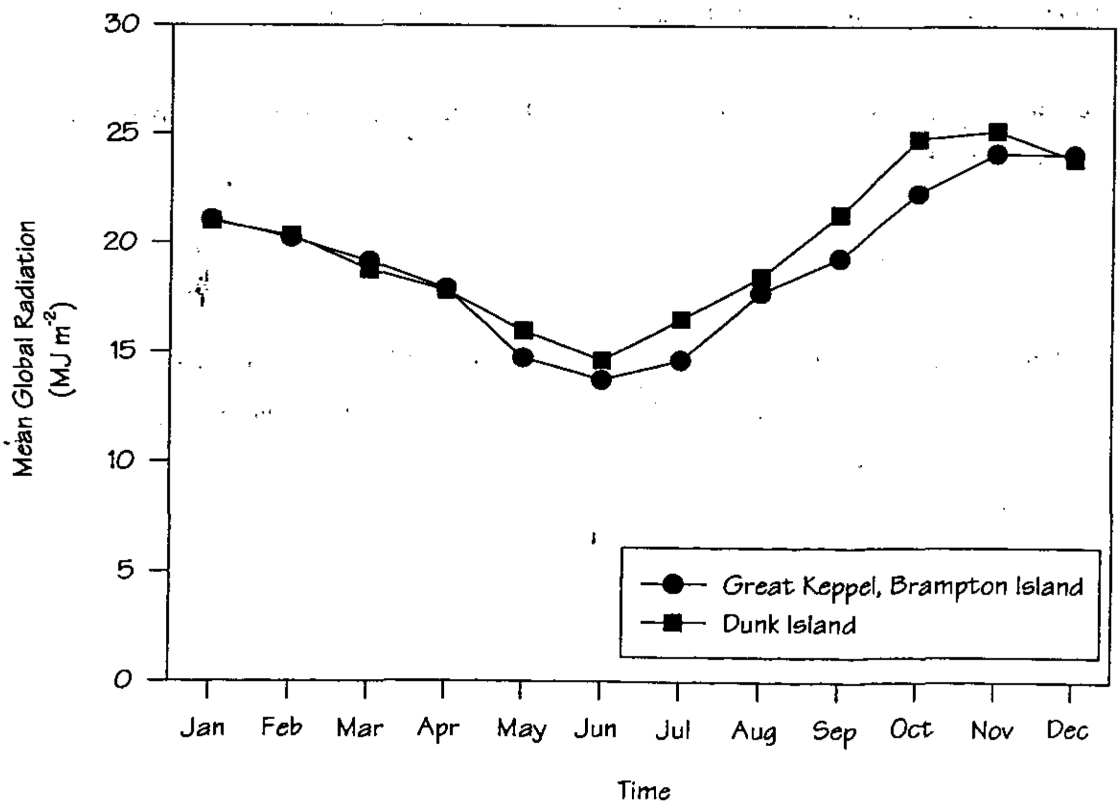

Figure 12. Mean monthly global radiation adopted for selected GBR resort islands.

3.1.2.3 Soil Hydraulic Properties

Undisturbed soil samples were collected from six representative plots within each resort golf course area. Soil samples were taken down to 1 m depth with a soil core sampler, and were divided into 8 different layers (0-5, 5-10, 10-20, 20-30, 30-50, 50-70, 70-85 and 85-100 cm). Laboratory measurements of the water retention data, saturated conductivity and porosity were made on two replicate samples according to standard procedures detailed in Klute (1982).

Calculation of the Brooks and Corey (1964) coefficients (λ, ψ_a and θ_r) for fitting moisture retention curves to the experimental data, was achieved by least squares fitting of the $\theta(\psi)$ relationship for tensions between 5% of air-filled porosity and 5 bar. A log-linear relation was assumed at higher tensions and a linear relationship at lower tensions. The wilting point was taken as the moisture content at a tension of 15 bar. Hysteresis effects were not accounted for.

The unsaturated conductivity function was then predicted using the procedure of Mualem (1976). Bypass or macropore flow through the profile was assumed negligible for both field sites. Tortuosity - which accounts for pore correlation and flow path, was set to 0.5 based on Mualem (1976) who demonstrated that such a value might hold as the best estimate.

The fitted water retention curves and predicted hydraulic conductivity curves are displayed in Figures 13 a,b to 15 a,b.

3.1.2.4 Initial and Boundary Conditions

The upper boundary condition was essentially defined by the surface runoff parameter a_{surf}, to account for the degree of runoff expected after the infiltration capacity of the uppermost soil was exceeded. It essentially provides a representation of the gradient or slope of the system under investigation.

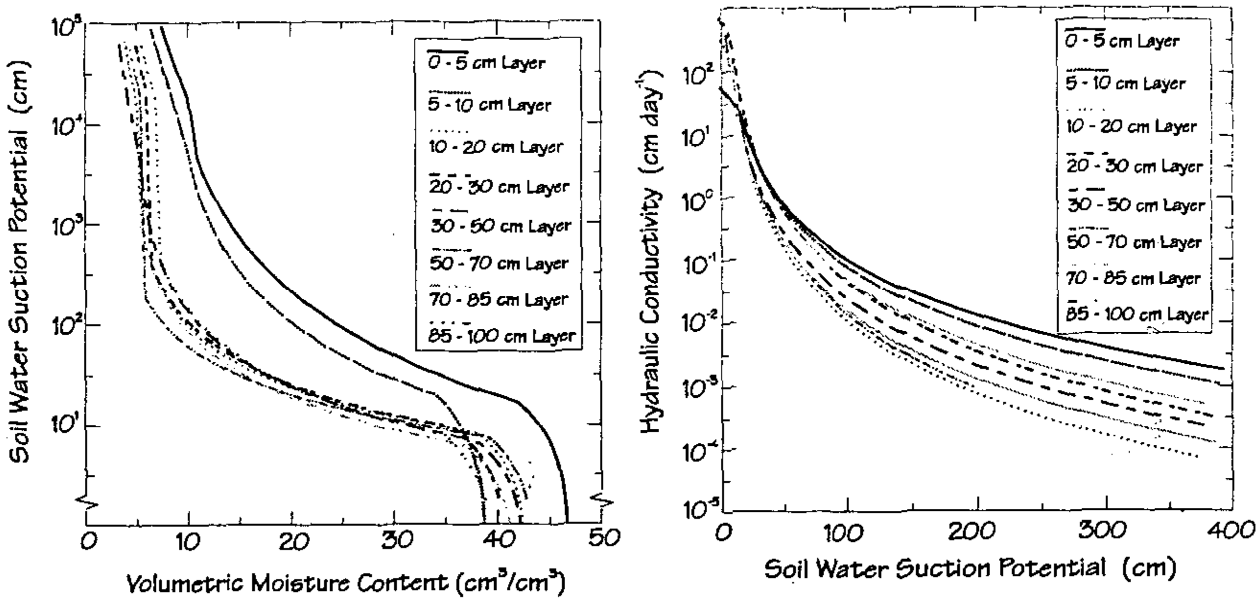

Figure 13. (a) Fitted soil water retention curves (left) and (b) predicted hydraulic conductivity curves using Mualem (1976) model with n=0.5 (right), for Great Keppel Island golf course.

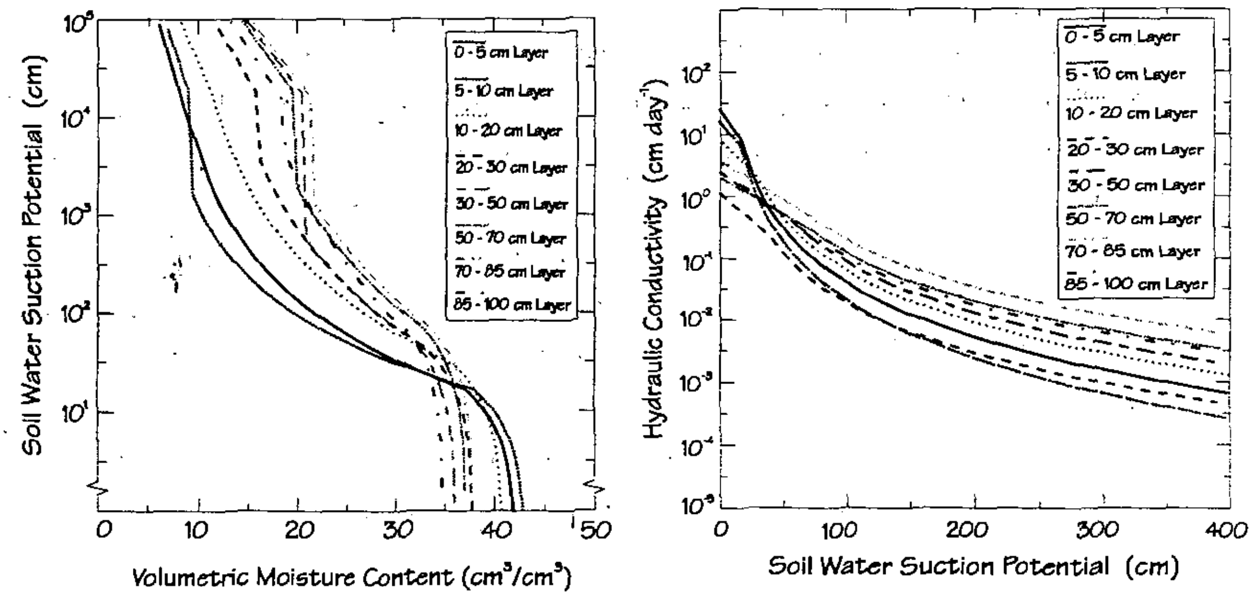

Figure 14. (a) Fitted soil water retention curves (left) and (b) predicted hydraulic conductivity curves using Mualem (1976) model with n=0.5 (right) for Dunk Island golf course.

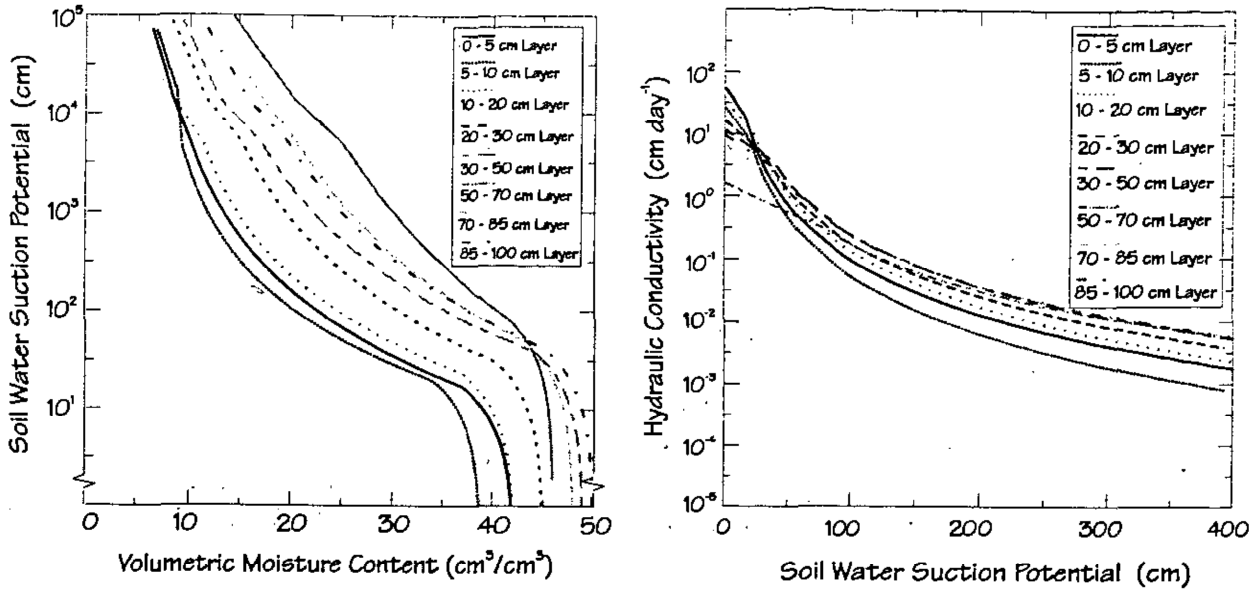

Figure 15. (a) Fitted soil water retention curves (left) and (b) predicted hydraulic conductivity curves using Mualem (1976) model with n=0.5 (right) for Brampton Island golf course.

Ephemeral surface runoff flow data were collected from a H-flume control structure and associated bubbler flow-meter located at a downstream reach of the Dunk Island golf course catchment for a period of approximately 12 months. Values of a_{surf} were trialed in the Dunk Island simulation for a series of representative rainfall events until a value of 0.6 was shown to generate total daily runoff (mm) of reasonable equivalence to the measured daily runoff flows. This value was also adopted for Brampton and Great Keppel Islands – which both possess a land gradient of between 2 and 8% in the area of investigation.

Initial values of soil water content at depths corresponding to those in the SOIL model simulations were taken from gravimetric moisture measurements of soil samples obtained from random sites within each island golf course.

Unsaturated flow throughout the soil profile was simulated for each island site given that groundwater measurements indicated water table depths in excess of 2 m, 20 m and 30 m below the Great Keppel, Dunk and Brampton Island golf courses respectively. In the absence of soil suction measurements in the lowest soil layer, the lower boundary condition was assumed as vertical water flow from the lowest compartment due to gravitational forces alone.

3.1.2.5 Turfgrass Properties

Turfgrass parameters are required to calculate the rate of water uptake from the profile through evapotranspiration. Values were largely based on previous studies of turfgrass as reported in the literature.

The maximum root depth was set to a constant value of 110 cm according to field observation, with a root distribution described by an exponential function. The reduction of plant water uptake due to drying soil is performed separately for each soil layer and begins when a critical tension in the soil is reached. Although the rooting density of vegetation growing in dense sands is generally not considered a limiting factor for water uptake (van Keulen et al. 1975), this can be the case in soils high in clay at high tensions, where roots do not penetrate the structural elements of the soil. In order to reflect the higher resistance of water transport to the root surface at high soil water tensions in the heavy clay soils of both Dunk and Brampton Island, the critical tension was set at 500 cm water, while a value of 1500 cm water was ascribed to Great Keppel Island.

The maximum standing height of cut turfgrass was estimated to be 3 cm. Displacement height and roughness length were assumed to be 70% and 10%, respectively of this stand height (Monteith and Unsworth, 1990).

The maximum leaf area index (LAI) is defined as the total quantity of leaf area present per unit area of soil surface. There is a decline in LAI as the cutting height is lowered; however, the increase in shoot density as the mowing level is lowered tends to buffer the reduction in LAI. A measured LAI of 3.2 for well-watered bermudagrass by Jalali-Farahani et al. (1994) was adopted here.

For the calculation of the surface energy balance, and in particular, the prediction of net solar radiation, the total reflectance of both diffuse and direct solar radiation from a plant surface (albedo) is required. A value of 0.26 for the albedo of lawn was taken from Robinson (1966) and assumed to remain constant for the whole simulation period.

The effective surface resistance of the vegetation, r_{sc}, used to calculate the potential transpiration, was taken to be 62.5 sm^{-1} from Jalali-Farahani et al's (1994) study of well-watered bermudagrass.

3.2 Sensitivity Study of Spatial Variability in Soil Hydraulic Properties

If soil water dynamics for a specific site are predicted based on a limited number of soil hydraulic data points, the results may not be representative of the area as a whole. Beckett and Webster (1971) reviewed data on the spatial variability of many physical properties of soils. They found that there was usually an appreciable variance over an area of 1 m^2, and their data showed that this variance increased roughly in proportion to the logarithm of the area. This can be compensated somewhat by calibrating a model so that measured average field water dynamics are reproduced.

Historically, numerous statistical methods have been used to account for spatial variation in soil properties. Geostatistics has often been employed to describe the spatial distribution of input and output variables and to calculate weighted areal means (Kutilek & Neilsen, 1994). Another technique involves accounting for spatial variability by scaling the water retention and conductivity data using the similar media concept (Miller & Miller, 1956).

To improve the understanding of the processes governing water flow and the subsequent leaching of NO_3 at each resort island site, spatial information on the soil properties was derived from the sampling regime described in Section 3.1.2.3. The mean (σ), standard deviation (SD) and the coefficient of variance (CV) of several measured soil hydraulic properties (θ_s, θ_{wilt}, K_{sat}) for each island are given in Appendix 1a-c.

Initially, areal arithmetic means of the measured soil properties were used in conjunction with least squares regressions of the measured $\theta(\psi)$ relationship using Brooks and Corey coefficients, to predict the deep percolation of water below the lowest layer to groundwater over a one-year time frame. Deep percolation was chosen as the output variable to be studied due to its obvious association with the leaching fraction predicted from SOILN. It should be noted that simple arithmetic spatial averaging of an input parameter is theoretically appropriate only if the equation is linear with respect to the parameter. However, even though unique relationships between different variables in this model are highly non-linear, the association between particular variables may be effectively linear as a result of the feedback mechanisms between different processes resulting in smoothing of nonlinear relations.

The influence of spatial variations in soil hydraulic properties on predicted deep percolation was then examined over a one-year period under natural conditions. This was undertaken by running the model for a matrix of 243 combinations to represent stochastic variations for specific soil properties. More specifically, a series of random numbers were generated which had equal probability of falling within the range specified by minimum and maximum values corresponding to one standard deviation above and below the mean, respectively. In the case of the Brooks and Corey coefficients, the range was determined by least-square regressions to each outlying set of the measured $\theta(\psi)$ relationship. Importantly, for each simulation, parameters characterising vegetation and meteorological conditions remained unchanged.

Envelopes of the variation in the accumulated deep percolation predicted over a one-year period compared to the accumulated deep percolation given by the area-averaged mean hydraulic properties are shown in Figures 16(a-b), 17(a-b) and 18(a-b) for Dunk, Great Keppel and Brampton Islands respectively.

The envelope of accumulated deep percolation predicted for Dunk Island is remarkably similar to that for Brampton Island. This indicates that simulations using mean hydraulic properties are more representative of the upper level of predicted accumulated deep percolation than the lower level. The Great Keppel Island results showed the least variation, a consequence of the low spatial variability in the soil parameters at this island. As expected, Brampton Island exhibited the greatest range of variation in accumulated deep percolation resulting from the more variable lithology observed there.

Overall, the assessment of the influence of spatially variable soil hydraulic parameters on water flux predictions suggests that deep percolation may vary with time depending on the proportions of dry and wet cycles at these sites. As expected, the spatial variation in deep percolation was more substantial during the wetter parts of each period and much less variable than the dryer periods. During a year with few wetting events, spatial variation in deep percolation may be minimal, however in more tropical areas such as Dunk Island, significant variations of deep percolation response are expected. Notably, the relevant output variables that generated each site-specific envelope of deep percolation were later used in all SOIL simulations and in the SOILN model for describing the upper and lower bounds of the NO_3^- leaching fraction.

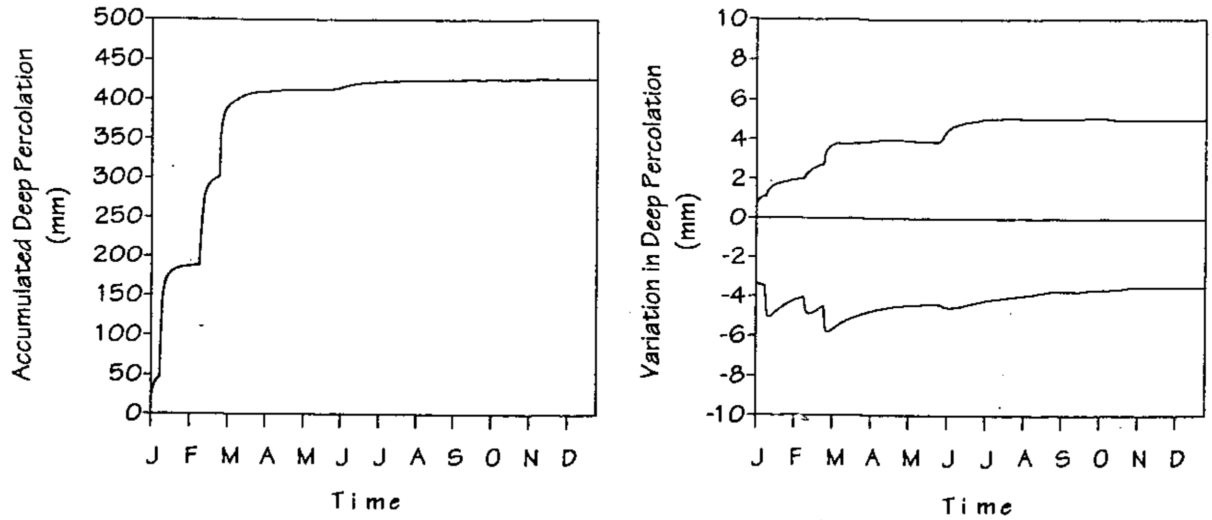

Figure 16. (a) Simulated accumulated deep percolation for Great Keppel Island using area averaged hydraulic soil properties (left). (b) Envelope of the variation in simulated accumulated deep percolation compared to (a) (right).

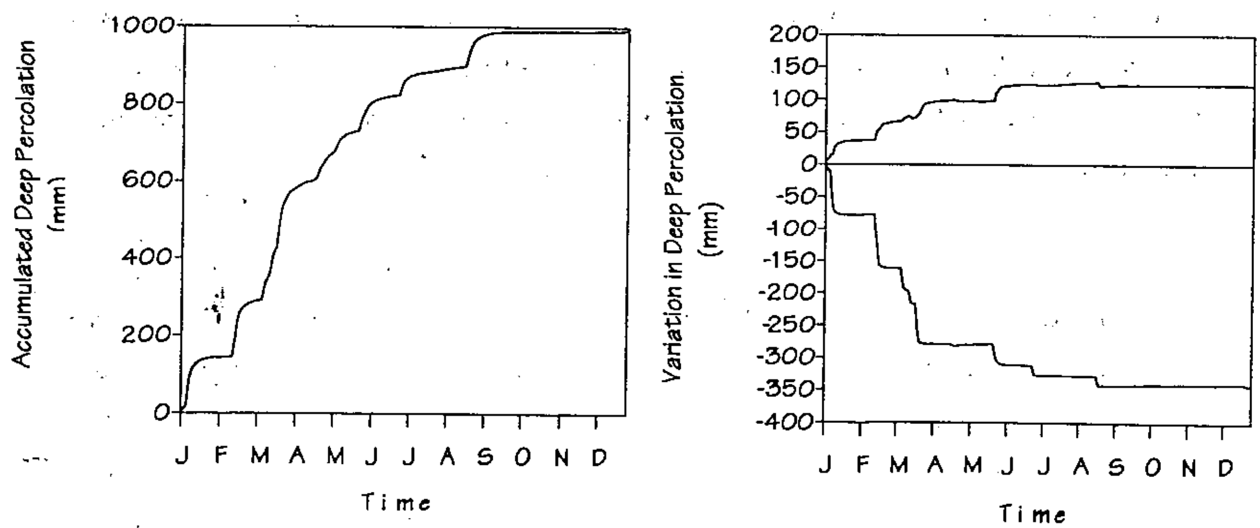

Figure 17. (a) Simulated accumulated deep percolation for Dunk Island using area-averaged hydraulic soil properties (left). (b) Envelope of the variation in simulated accumulated deep percolation compared to (a) (right).

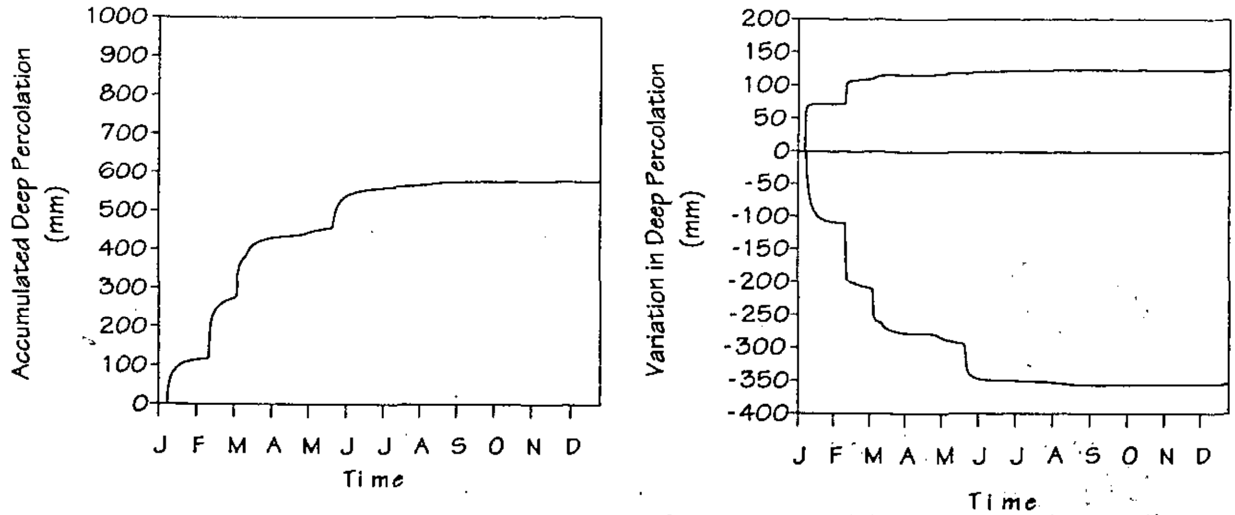

Figure 18. (a) Simulated accumulated deep percolation for Brampton Island using area-averaged hydraulic soil properties (left). (b) Envelope of the variation in simulated accumulated deep percolation compared to (a) (right).

3.3 The SOILN Model

The model SOILN (refer to Figure 19) includes the major processes involved in soil N transformations and transport and was first presented and described in detail by Johnsson et al. (1987).

The soil profile is divided into layers based on physical and biological characteristics. Both inorganic and organic N pools are represented within each soil layer. Organic N is divided into two pools classified as litter (undecomposed crop residues, dead roots and microbial biomass) and humus. The litter N pool is coupled to corresponding pools of carbon to control the rate of mineralisation and immobilisation. The inorganic (mineral) N pools consist of NH_4^+ and NO_3^- whereas N residing in living aboveground and subsurface plant tissues is combined into a single plant pool.

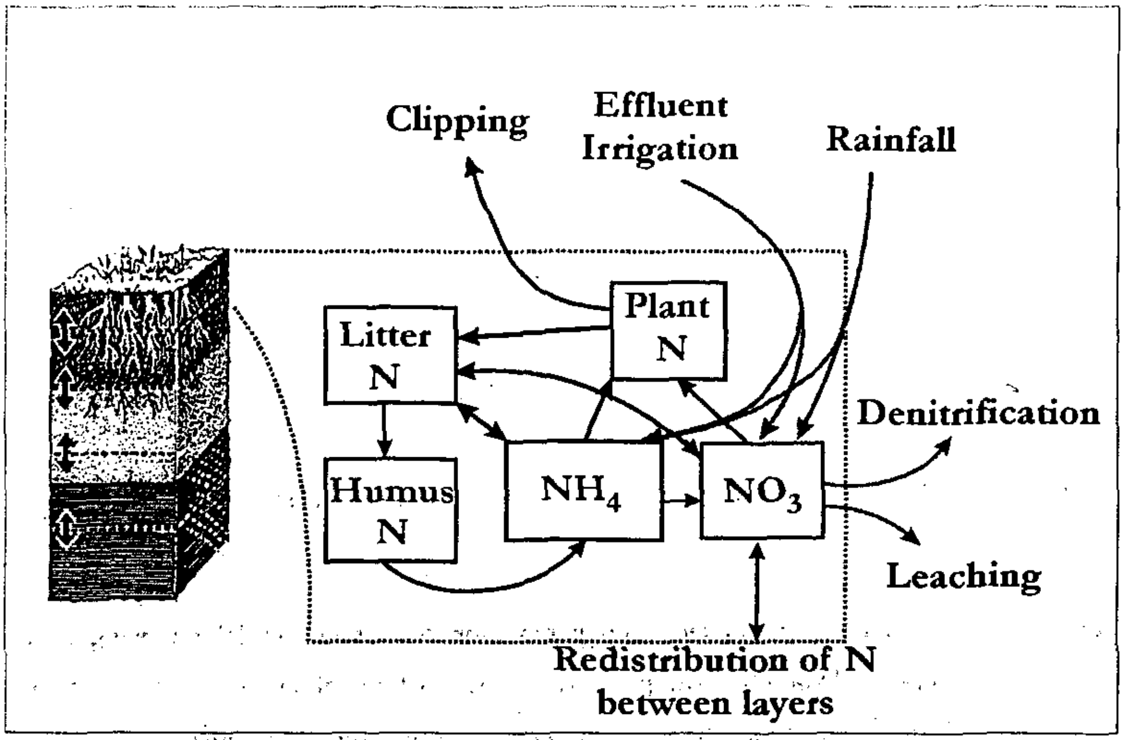

Figure 19. Nitrogen flow and storage simulated by the SOILN model (adapted from Eckersten et al.,1996)

External inputs of N to the uppermost soil layer include effluent irrigation and rainfall while outputs of N can occur from each soil layer by denitrification or by plant uptake. Nitrate can also be transported between the soil layers or in deep percolation from below the soil profile and is calculated as the product of water flow and NO_3^- concentration in the soil layer from which the water flow originates. The NO_3^- concentration is calculated as the total storage of NO_3^- divided by the liquid water storage in each soil layer. It is assumed that NH_4^+ does not move with water flow in the soil. It should also be emphasized that diffusion and dispersion are not explicitly accounted for, although partitioning the soil into discrete layers of variable thickness partnered with the finite difference representation of convection of N results in numerical dispersion.

3.3.1 Selected Input Parameters

The daily input or 'driving' variables required for modelling with SOILN included simulated water content and temperature in each soil layer, water flow between soil layers, overland runoff, global radiation, ambient temperature, ratio of actual to potential water uptake and surface infiltration. This data was acquired by transferring the output from prior simulations generated from SOIL, given that discretization of the soil profile matched for both models.

Furthermore, the selection of specific parameters for simulating N cycling and transport processes in SOILN was based on an extensive review of literature to obtain values representative of turfgrass environments in sub-tropical conditions for the broad soil classifications of each of the sites being studied. The reliability of the predicted outcomes from SOILN are dependant on the of parameter values describing individual processes, so particular attention was given to the choice of parameters governing the rate of N transformation processes in the soil and turfgrass growth descriptors. In several instances, literature pertaining to such conditions could not be adequately sourced and available literature was restricted to forage grass systems in milder, less temperate conditions. To test the adequacy and applicability of the model, particularly in light of such parameter deficiencies, a sensitivity study was performed on key parameters related to the biotic processes and interactions in SOILN.

Some parameters adopted for this study were those originally derived by the authors of SOILN and are listed in Appendix 3. These principally related to soil abiotic processes including soil water content and temperature response functions. Only parameters which differ from those presented in earlier studies by the authors using SOILN, or where the literature review has lent additional support to existing parameter values, are commented upon in detail in the sections below and also appear in Appendix 3. Importantly, no additional adjustments or tunings were intended for parameter values to improve model fit due to the inherent uncertainties of many of the input parameters and the complexity of their interactions, particularly given the likely limitations of field data available for validation. A comprehensive sensitivity analysis (refer Section 4.3) was preferred for establishing the applicability and relevance of the assumed SOILN modelling parameters.

3.3.1.1 Initial Conditions and External Inputs of N

As interest in state variables for soil N and C pools focused only on accumulated changes of these pools over a long simulation period and not in short-term fluctuations, initial values were estimated from literature with the exception of the humus N pool (refer Appendix 4).

Mean values for organic (humus) N for each of the eight soil layers were obtained from laboratory measurements of the soil N organic matter for 6 sets of cored soil samples taken from within the Brampton Island golf course environs.

The level of NO_3^- and NH_4^+ in the soil layers was based on measured quantities per unit mass from a vertisol in southeast Queensland by Probert et al (1998) and converted to mass per unit area by applying a typical bulk density corresponding to that measured at Brampton Island. Initial values of litter C and N in the uppermost four layers were taken from work by Robertson et al. (1993) on C and N availability in subtropical Queensland soils under grassland, and again converted to mass per unit area as required by SOILN.

The initial quantities of grass tissue biomass in root, stem and leaf at the beginning of each simulation were based on data given in Topp and Doyle (1996), with the initial values of N partitioned in the leaf, stem and roots extracted from Wilman et al. (1994).

Simulations were run for two years prior to the start of the period of study to limit the effect of errors in the assumed initial conditions. The initial conditions and reference source of biomass, C and N pools assumed across all three study sites are given in Appendix 4.

Wet atmospheric deposition of N was calculated from net daily rainfall and a mean concentration of mineral N in the precipitation. The concentrations of the various mineral N species in rainfall were adapted from analyses of the dissolved nutrients in rainfall collected throughout the GBR and western Coral Sea by Furnas et al. (1994) given in Table 5.

Table 5. Dissolved N in GBR rainfall (adapted from Furnas et.al., 1994).

NO_3^-				NH_4^+			
µg NO_3 L^{-1}		as µg N L^{-1}		µg NH_4 L^{-1}		as µg N L^{-1}	
Mean	Median	Mean	Median	Mean	Median	Mean	Median
1.31	0.36	0.3	0.08	3.31	2.01	2.7	1.56

The median concentration was selected for modelling purposes in order to diminish the importance of extreme values in the sample data, consequently the mineral N concentration of rainfall (DEPWC) and the fraction of NH_4^+ in rainfall (DEPNH4W) were set to 1.6E-3 and 0.95 respectively.

In order to specify effluent irrigation in SOILN, inputs of N were given as quantities of fertiliser (kg N ha^{-1}) divided into NO_3^- and NH_4^+ components and ascribed a dissolution rate of 100%. Regularly measured data for total mineral N concentration and speciation in sewage effluent were available for use in the SOILN model (refer to Figures 3, 5 and 7). The N data were linearly interpolated between sampling times to a daily time scale and extrapolated to the period before and after the 4-year period of effluent sampling. The mean annual applied TN for each island under various effluent-loading regimes is shown in Table 6.

Table 6. Effluent irrigation mean TN loading rates for SOILN modelling

Fraction of total daily effluent irrigated [%]	Great Keppel Island Mean Applied TN [kg N $ha^{-1}yr^{-1}$]	Dunk Island Mean Applied TN [kg N $ha^{-1}yr^{-1}$]	Brampton Island Mean Applied TN [kg N $ha^{-1}yr^{-1}$]
25	185	43	89
50	370	86	178
75	555	129	267
100	740	172	356

3.3.1.2 Denitrification

SOILN considers denitrification a zero-order process based on a daily potential rate that is modified by response functions for temperature, water content and NO_3^- concentration. The equations governing denitrification in SOILN are fully outlined in Johnsson et al. (1987).

Reported rates of denitrification in Australian soils vary considerably, although conditions of soil core incubation also vary, thus limiting comparisons of potential denitrification. Catchpoole (1975) determined that 27% of applied fertiliser N was denitrified when a grass pasture established on '*prairie-like soil*' was waterlogged. Pu et al. (1999) measured the denitrification occurring in grey cracking clays supporting a variety of grass species and found losses between 1.2 and 1.8 kg N ha^{-1} d^{-1} depending on the level of added residues. They also reported that most of the N loss occurred in the top 0.5 m, a result that was further supported by Burford and Bremner (1975). Avalakki et al. (1995) measured nitrous oxide (N_2O) emissions from denitrification on a black earth (Vertisol) and detected emissions ranging from 0.38 - 0.95 kg N ha^{-1} d^{-1} for incomplete saturation and 1.3 - 2.4 kg N ha^{-1} d^{-1} for complete saturation.

A number of overseas studies have also attempted to ascertain levels of N_2O emission from below various soil and vegetative systems. However, it should be noted that N_2O could also have been formed during the oxidation of NH_4^+ during the nitrification process. A more comprehensive examination of the effect on denitrification of soil depth between 0 and 50 cm under grazed grass plots was performed by Ryan et al. (1998). Denitrification in the 0 to 10 cm layer was shown to be much greater than in the lower layers, with only 20% of the total 16.5 kg N ha^{-1} yr^{-1} denitrified at greater depth.

Also, the seasonal variation in annual denitrification rate has been measured at between 1.2 and 1.3 kg N ha^{-1} d^{-1} in a waterlogged sandy soil grassland meadow over a three year period (Davidsson and Leonardson, 1997).

Luo et al. (1998) also examined the influence of soil depth on soil denitrification activity of both sandy loam and silt loam below pasture. Maximum denitrification activity was determined in the surface soil (0-5 cm) which decreased exponentially with depth, regardless of soil type or time. Accordingly, an exponential distribution of denitrification below the soil surface was adopted for SOILN simulations at all three sites.

Therefore, the daily potential denitrification rates (DENPOT) for both Dunk and Brampton Island were set to 1.5 kg N ha^{-1} d^{-1}, with 1 kg N ha^{-1} d^{-1} selected for Great Keppel Island. Such values were chosen to correspond with the broadly clay and sandy soils exhibited respectively at these islands. In addition, the assumed depth where the denitrification capacity ceased (DENDEPTH) was ascribed a value of 0.5 m.

The influence of temperature on denitrification is allowed for in SOILN by a Q_{10} expression – effectively representing the multiplicative change in denitrification rate when a 10^0C increase in temperature.

Smith et al. (1998) investigated the effect of a variation in soil water content and temperature on the denitrification response from clay loam and sandy loam with vegetation, at or above field capacity. Their study demonstrated that when soil mineral N was non-limiting, exponential relationships existed between the N_2O flux and both moisture content and temperature. The Q_{10} for the response ranged between 5 and 15 for a clay loam soil with turf, while in a sandy loam with turf, a Q_{10} of 1.6 was deduced.

They also indicated that the higher Q_{10} value for clay loam was attributed to an enhanced respiratory demand of oxygen and an associated expansion and prevalence of anaerobic zones. This increased anaerobic volume for a given soil volume had previously been shown to raise the Q_{10} for denitrification (Smith, 1997). Thus, based on a broad classification of the soil type found at the GBR resort islands under study, a Q_{10} (TEMQ10D) value of 10 was adopted for Brampton and Dunk Islands and a Q_{10} of 1.6 was chosen for Great Keppel Island.

The water response is limited to a range close to saturation where denitrification increases with increased water content to a maximum level at saturation. Smith et al. (1998) found that the total N_2O emissions from cut grasslands increased sharply as the soil water content increased. Moreover, for a clay-loam soil, an exponential increase in emissions was evident as the moisture content rose from 50% to maximum N_2O emissions at 90% of porosity. Consequently, the shape parameter for soil moisture effect on denitrification (DEND) was specified as a non-linear response function with an assumed exponent of 2.

The critical air-filled porosity at which denitrification was accelerated for a perennial ryegrass sward established on a fine sandy loam in New Zealand was 17% (Ruz-Jerez et al., 1994). The water content interval (MOSDEN) defining increased denitrification activity from zero to optimum activity was set to 15% for each site in the absence of further information.

The NO_3^- content of the soil was identified as a limiting factor for denitrification activity at some depths, particularly in surface soils where the concentrations were low (Luo et al., 1998). SOILN therefore calculates the NO_3^- concentration response from a *Michaelis-Menton* expression with a half-saturation constant (DENHS) i.e. the concentration at which the rate is 50% of the maximum if all other conditions are optimal. This value was assumed equal to 10 mg N L^{-1} (Johnsson et al., 1991).

3.3.1.3 Turfgrass Growth, Uptake and Redistribution

Lawn was assumed to be cut at regular intervals of 14 days, with 80% of leaves remaining after mowing. The C and N in leaf, stem and root residues then pass into the litter pool. The grass C and N are each divided into three fractions, harvested output and above- and below-ground residues. Above-ground residues are incorporated into the litter pools of the upper soil layer together with a corresponding amount of carbon calculated from the leaf and stem C:N ratio - the relative quantities of each determining the C:N ratio of the litter pool. Below ground residues are partitioned into the litter pool of each soil layer together with the quantities of C and N corresponding to the root pool on the last day of growth.

The methodology of SOILN regarding plant N uptake assumes NH_4^+ and NO_3^- are equally available to the grass and calculated according to the relative amounts of the two ions available. Potential daily turfgrass growth per unit area of soil surface is proportional to the light intercepted by the canopy deduced from the global radiation, radiation extinction coefficient and leaf-area index according to Beers' law. This is subsequently converted into biomass by multiplying by a potential radiation use efficiency coefficient. The predicted daily growth is then found by applying reduction factors to the potential growth to account for low soil temperatures as well as moisture and mineral N deficits.

The grass N demand is a function of the daily biomass formation in the different plant tissues and is determined by a maximum N concentration in the respective plant tissues. A fraction of the total soil mineral N is also available for uptake by the grass. The modelled N uptake is the lower value of the grass N demand and the available amount of mineral N. More detail on the SOILN crop growth sub-model can be found in Eckersten and Jansson (1991).

To adequately parameterize the SOILN model for turfgrass photosynthetic biomass allocation, leaf assimilation, respiration and clipping redistribution, a wide range of literature relating to various grass types was consulted.

A study of irrigated bermudagrass by Devitt (1989) observed a decrease in root length density with depth, with the highest values occurring in the 0 - 5 cm layer for sandy loam, while a shallower root system was observed for a clay soil. Approximately 75% of the root systems were located in the upper 57, 55 and 40 cm of the soil strata in the sandy loam, silt loam and clay soils respectively. The lowest level of roots (ROOTDMIN) was observed at approximately 1.1 m. To broadly reflect these outcomes, an exponentially decreasing distribution was applied to root density below the soil surface, with the parameter determining root depth as function of root biomass (ROOTDINC) given a value of -0.06 for Great Keppel Island and -0.04 for both Brampton and Dunk Islands.

The maximum proportion of soil mineral N available for turfgrass uptake at each daily time step of the model (UPMA) was selected as 8% of the total mineral N pool for each soil layer from Johnsson et al. (1987).

When the plant demand was higher than UPMA, a compensatory uptake from the other layers (UPMOV) of 1 (=100%) was calculated. Such a high proficiency of grass root systems to reallocate the source of mineral N for uptake was supported by a modelling study of N uptake of perennial ryegrass cover (Blomback and Eckersten, 1997).

A special option where the turfgrass leaf assimilation rate was a function of a light response curve for a single leaf type integrated over the canopy was implemented. Leaf assimilation was calculated using a light response curve for photosynthesis taking account of growth respiration (France and Thornley, 1984). The following input parameters for this option were specified from a literature review of grassland systems (refer Appendix 3): maximum leaf photosynthesis rate at optimal temperature (PPMAX20(1)), rate of decline of maximum leaf photosynthesis rate with increased leaf area index (PPMAX20(2)), leaf transmission coefficient (PTRANSM), and respiration growth efficiency (PGRESP).

Similarly, parameters required to define the radiation use efficiency (PHOEFF) for the grass canopy were based on Hodgkinson et al. (1989) who undertook a light conversion analysis of two perennial tussock grasses grown in south-east Queensland pastures. Light utilising efficiency was shown to be similar throughout periods of regrowth following either grass cut weekly or cut infrequently, with a value of 3.5 g DM MJ^{-1} representative of regularly mowed turfgrass. In addition, the light extinction coefficient (EXTCOEF) for warm season switchgrass ranged from 0.57 to 0.72 (Madakadze et al., 1998), therefore a value of 0.6 was considered appropriate for this modelling exercise.

The leaf N concentrations at which maximum and minimum photosynthesis occur (NLEAFXG, NLEAFN) were set to values based on studies of N concentration in grass clippings (Hodgkinson et al., 1989; van Keulen et al., 1989).

With respect to the parameters controlling the biomass allocation in turfgrass, the specific leaf area (WLAI) which is defined as the leaf area per dry weight of leaf, was deduced from a study by Boot and den Dubbelden (1990) of two perennial grass species on inland dunes. WLAI varied approximately between 0.018 and 0.032 m^2 gDM^{-1} depending on the species and rate of N treatment. The minimum value of 0.018 m^2 gDM^{-1} was adopted for this study, corresponding to the narrower, non-rhizomatous species with a highly availability N supply, as expected for a turfgrass in an effluent irrigated environment.

The proportion of growth allocated to plant parts was measured by Hodgkinson et al. (1989), with the fraction of total growth apportioned to roots (AROOTN) relatively constant at between 0.05 and 0.1 d^{-1} for both grass species during regrowth after cutting. Coefficients for leaf area development as functions of shoot biomass (ALEAF(1) and ALEAF(2)) were taken from leaf:stem ratio experiments conducted by Wu et al. (1998) on grassland fields in Scotland.

Also, the daily fraction of leaf and stem biomass lost to litter (ALITTERL and ALITTERS) were prescribed the values 0.026 and 0.023 d^{-1} respectively, on the basis of perennial ryegrass measurements by Sheehy et al. (1980). Similar work by Thornley and Verberne (1989) on grasslands revealed that the daily fraction of root biomass lost to litter (ALITTERR(2)) was of the order of 0.03 d^{-1}.

3.3.1.4 Mineralisation / Immobilisation and Nitrification

Nitrogen mineralisation is defined as the biological decomposition of organic materials in soils and their conversion to the inorganic forms: NH_4^+ and NO_3^-. The balance between the C and N content of the energy sources dictates whether organic N is mineralised to a plant available form or residual soil N is immobilised by microorganisms. (Schepers and Mosier, 1991). More precisely, if the C:N ratios of the above-ground litter are greater than the humus pools, the redistribution of residues into the system will often cause net immobilisation due to the high demand of microbial growth during decomposition. Conversely, if the litter pools become dominated by recycled microbial biomass with a small C:N ratio, net mineralisation will result.

Mineralisation of humus N is calculated in SOILN as a first-order rate process controlled by a specific mineralisation constant and response functions accounting for the influence of soil temperature and soil moisture. Corresponding N flows are calculated assuming a C:N ratio of decomposer biomass. When net immobilisation occurs, the immobilisation rate is limited to a maximum fraction of the mineral N content in the soil. Importantly however, although NH_4^+ and NO_3^- can both be immobilised, NH_4^+ is assumed to proceed first. Specific algorithms detailing the mineralisation/immobilisation transformations in soil and the nitrification of NH_4^+ to NO_3^- are described in detail by Johnsson et al. (1987).

Schepers and Mosier (1991) estimated that approximately 2% of the total organic N in surface soils were mineralised annually. However, if irrigation was applied regularly, the mineralisation of large amounts of recently added grass residues could slowly promote the organic matter pool and factor this rate by as much as twice. They also warned that estimates of mineralisation should be viewed with an uncertainty of ± 25 - 50%.

Two tropical Australian soils, a clay loam and sand, were shown to undergo 4.8% and 12.5% mineralisation of organic N respectively in the first year after clearing, which then decreased to 5 - 5.9% in the years following (Wetselaar, 1967). Bremner (1965) also concluded that most of the organic N in soils was resistant to biological attack and < 3% of the total organic N was mineralised annually.

A rate of humus mineralisation (HUMK) of 1.5E-5 d^{-1} was chosen based on the range prescribed by Jansson and Andersson (1988), whereby values from 1.0E-5 - 2.1E-5 were successfully applied to N modelling of a cereal-dominated watershed of clay till in southern Sweden. These values were further supported by Katterer and Andren (1996), who calibrated HUMK to a value of 3.0E-5 d^{-1} when modelling N movement in a clay soil supporting a grass ley subjected to daily irrigation and fertilization.

Probert et al. (1998) modelled the N dynamics of a wheat growing vertisol at Warra in south-east Queensland and an alfisol at Katherine, Northern Territory supporting legume leys, both fertilised at 0 - 75 kg N ha^{-1}. Parameters successfully applied in their modelling study included a litter specific decomposition rate (LITK) of 8E-3 d^{-1}, efficiency of internal synthesis of microbial biomass in litter (LITEFF) of 0.4 and the fraction of N and C in above ground residues converted to litter (ABOVEK) of 0.1 d^{-1}.

As the predictive ability of the model in terms of NO_3^- migration proved satisfactory across the two data sets in Probert et al. (1998), such values were considered appropriate for defining C and N flows between the litter and humus pools in SOILN simulations for this study. Notably, Duble and Weaver (1974) reported that decomposition of leaf and stem tissue was twice as fast as that of root tissue, however due to the fact that root matter and leaf residue were both allocated to the same litter pool, only one value for ABOVEK could be applied.

Another important modelling parameter for mineralisation and immobilisation determination is the C:N ratio of micro-organisms and humified products (CNORG). Values ranged from a mean of 5.2 in the top 28 cm of grassland soils in subtropical Queensland measured by Robertson et al. (1993) to a value of 8 used for soil N modelling in south-east Queensland soils in Probert et al. (1998). A CNORG value of 6 was therefore chosen for this work. This value is also within the range 5 - 15 found in other studies on microbial biomass C:N ratios (Grace et al., 1993; Bloemhof and Berendse, 1995).

With respect to the C content of grass biomass lost to litter (CPLANT), the total leaf C concentrations of three tropical pasture grasses grown in Australia fell within a narrow range of 0.4 - 0.47 g C g DM^{-1} (Ghannoum and Conroy, 1998). A value of 0.45 g C g $DM^{-1}$was therefore utilised for SOILN modelling.

Specification of the parameter controlling the litter C humification fraction was based on a study of the decomposition and N mineralisation of above-ground plant material in two unfertilised grassland systems (heavy clay and loamy sand) in the Netherlands (Bloemhof and Berendse, 1995). The litter C utilisation efficiency for dead grass litter (LITHF) was found to be 0.35. Given that reported values have ranged from 0.2 - 0.4 in other models of decomposition in grassland systems (De Ruiter et al., 1993; Van Veen et al., 1985), a LITHF of 0.35 was considered appropriate for this study.

Nitrification of NH_4^+ to NO_3^- in SOILN is calculated as a first-order rate process, modified by the excess of NH_4^+ when the NH_4^+: NO_3^- ratio exceeds an assumed equilibrium for the soil. The transfer rate of nitrification is further defined by a specific daily rate constant (NITK) and the temperature and moisture response functions previously discussed. A NITK range of approximately 0.03 - 0.09 d^{-1} was derived from field experiments involving various fertiliser applications to cereal crops in Queensland, with higher values occurring in more moist soil conditions (Strong and Cooper, 1992). Thus, given that relatively high soil moisture contents would be anticipated from daily effluent irrigation, a NITK value of 0.08 d^{-1} was selected.

4. RESULTS

4.1 SOIL Simulations

SOIL was used to simulate upper and lower bounds of the accumulated water balances at each island for different irrigation loads, based on the previously defined range of soil spatial variability (refer to Tables 7, 8 and 9).

Table 7. Accumulated water balance (mm) for the various hydraulic loading scenarios at Great Keppel Island over a 20-year simulation period.

Source/Sink	Hydraulic Loading (mm)									
	0		3.3		6.7		10		13.3	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Rainfall	23030	23030	23030	23030	23030	23030	23030	23030	23030	23030
Irrigation	0	0	24110	24110	48950	48950	73050	73050	97170	97170
Soil Evap.	6453	6368	9683	9730	10020	10020	10020	10020	10020	10020
Surface Runoff	0	0	0	0	0	0	0	0	0	0
Turfgrass Uptake	8607 (0.374)	8552 (0.371)	20287 (0.430)	20200 (0.429)	22860 (0.318)	22870 (0.318)	22760 (0.237)	22870 (0.238)	22860 (0.190)	22860 (0.190)
Deep Percolation	8136 (0.353)	8285 (0.360)	17300 (0.367)	17340 (0.368)	39200 (0.545)	39210 (0.545)	63390 (0.660)	63300 (0.659)	87390 (0.727)	87400 (0.727)
Δ Storage	-166	-175	-130	-130	-100	-120	-90	-110	-70	-80

Note: Values in brackets equal fraction of the combined rainfall and irrigation.

Table 8. Accumulated water balance (mm) for the various hydraulic loading scenarios at Dunk Island over a 20-year simulation period.

Source/Sink	Hydraulic Loading (mm)									
	0		1.0		2.0		3.0		4.0	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Rainfall	58260	58260	58260	58260	58260	58260	58260	58260	58260	58260
Irrigation	0	0	7310	7310	14610	14610	21920	21920	29220	29220
Soil Evap.	7563	7298	10030	9700	12060	11570	13190	12600	13880	13380
Surface Runoff	21270	9495	22550	10070	23860	10630	25420	11240	27190	11920
Turfgrass Uptake	16647 (0.286)	17642 (0.303)	18270 (0.279)	19290 (0.294)	19980 (0.274)	21060 (0.289)	21450 (0.268)	22550 (0.281)	22270 (0.255)	23370 (0.267)
Deep Percolation	12660 (0.217)	23670 (0.406)	14590 (0.223)	26340 (0.402)	16830 (0.231)	29450 (0.404)	19970 (0.249)	33630 (0.419)	24010 (0.274)	38650 (0.442)
Δ Storage	+120	+155	+130	+170	+140	+160	+150	+160	+130	+160

Note: Values in brackets equal fraction of the combined rainfall and irrigation.

Table 9. Accumulated water balance (mm) for the various hydraulic loading scenarios at Brampton Island over a 20-year simulation period.

Source/Sink	Hydraulic Loading (mm)									
	0		0.9		1.8		2.8		3.7	
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Rainfall	31550	31550	31550	31550	31550	31550	31550	31550	31550	31550
Irrigation	0	0	6570	6570	13150	13150	20450	20450	27030	27030
Soil Evap.	12140	9775	15170	12010	19540	15710	24030	19200	27540	21310
Surface Runoff	5418	259	6131	321	6555	343	7004	376	7421	385
Turfgrass Uptake	6910 (0.219)	7075 (0.224)	8100 (0.212)	8550 (0.224)	8600 (0.192)	9410 (0.211)	9200 (0.177)	10400 (0.20)	9830 (0.168)	11370 (0.194)
Deep Percolation	6992 (0.222)	14390 (0.456)	8624 (0.226)	17180 (0.451)	9895 (0.221)	19170 (0.429)	11660 (0.224)	21970 (0.423)	13670 (0.233)	25450 (0.434)
Δ in Storage	+90	+51	+95	+59	+110	+67	+106	+54	+119	+65

Note: Values in brackets equal the fraction of the combined rainfall and irrigation.

The mean predicted evapotranspiration (ET) for all cases during the study period ranged between 2.1 and 4.5 mm daily at Great Keppel Island, 3.3 and 5.1 mm d^{-1} at Dunk Island and 2.6 and 4.6 mm d^{-1} at Brampton Island. These values compare favorably with other studies of turfgrass ET (Beard, 1985; Carrow, 1991, Tovey et al., 1969). Additionally, the major proportion of the ET for both locations was shown to be from grass water transpiration, which is a realistic outcome in view of the significant coverage of the soil by turf.

As anticipated, the most significant surface runoff flows were simulated for Dunk Island. This was a consequence of the combined high rainfall and low soil conductivity, which resulted in a maximum accumulated runoff component of 31% of the total water input. In contrast, no overland flows were predicted for Great Keppel, a consequence of the highly permeable sand formation.

Based on the quantity of turfgrass uptake over the 20 year period at Great Keppel Island in particular, it appears that a maximum value (22 860 mm) was reached during the various cases of hydraulic loading. This effectively represents the maximum possible utilization of water by grass under the physical conditions at this site, which would have been initially reached at an irrigation rate somewhere between the relatively wide range of 3.3 and 6.7 mm d^{-1}. No such limiting value was attained with certainty for the series of hypothetical hydraulic loads at Brampton and Dunk Islands.

Simulated deep percolation of water from the Great Keppel and Dunk Island profiles increased with progressively higher quantities of irrigation. The former rose markedly, ranging from 35.3% to 72.7% of total water input, while a more moderately increasing range of possible values was indicated for Dunk Island. Brampton Island maintained a relatively constant range of percolation flowing to groundwater, of 22.2% to 45.6%.

At any point in each soil drying cycle, the amount of water that has been removed below field capacity represents the amount of soil water storage that must be refilled before any increase in percolation can occur. The simulated differences in percolation can be attributed to the larger volume of water that can be stored temporarily above field capacity for a clay-based soil. Therefore, water stored above field capacity can then be largely removed by turfgrass, even for the first day following high rainfall at the Dunk and Brampton Island sites.

The envelope of daily volumetric water content for the bottom of the soil profiles at each site is presented in Appendix 2 (a-e). For Great Keppel Island, the predicted soil moisture at 0.85-1.0 m depth of each site increases to an almost constant value as the irrigation rates increase. This implies that the soil experienced virtually no drying periods and is always at or close to field capacity. Significantly larger fluctuations in moisture contents for the corresponding soil layer at both Brampton and Dunk Islands are evident, demonstrating the influence of high rainfall events in these instances.

4.2 SOILN Simulations

A key result of the SOILN simulations was that frequent cutting of the leaf biomass and the resultant redistribution of the N and C to the aboveground organic pool have indirectly promoted the N available for further plant uptake and leaching. The cuttings were effectively another source of N to the system in an organic form which could directly mineralise to NH_4^+ or immobilise to soil humus (for later mineralisation to NH_4^+) depending on the soil C:N ratio. The influx of N from returned grass clippings provided up to an additional 60% of the initially applied effluent N to the system over a twenty-year period. The annual addition of clippings on sports turf has been estimated as 2000 kg ha^{-1} (Riem Vis, 1981), which contributes approximately 80 kg N $ha^{-1}yr^{-1}$ assuming a grass N content of 4% by weight. This compares favorably with a range of 22 to 221 kg N $ha^{-1}yr^{-1}$ across all sites and cases simulated.

The simulated uptake of N by turfgrass was thus shown to exceed the applied effluent N in all cases of wastewater application for Dunk and Brampton Islands (refer to Figure 20). The level of turfgrass N uptake predicted for Great Keppel Island was similarly high; exceeding applied effluent N when 25% of total daily sewage effluent was irrigated and ranging between 62% and 96% for other simulated cases.

These outcomes compare favourably with many investigations on the proportion of applied N taken up by grassland systems in Australia, when both the difference in N inputs and the tendency for harvesting of biomass in these studies, are taken into account. Two such studies (Henzell, 1963; Henzell, 1971) were undertaken on various pasture grasses growing in red podzolic (sandy loam) soils at Samford in southeast Queensland. Henzell (1963) reported the N recovery in mixed pasture grasses at 47% for an N input of 448 kg $ha^{-1}yr^{-1}$, while his examination of the N uptake by Rhodes grass from four fertiliser rates yielded plant recoveries as high as 62% at 224 kg N ha^{-1} yr^{-1} (Henzell, 1971).

Also, Lazenby and Lovett (1975) investigated the uptake of N by five pasture grass species grown on the northern tablelands of New South Wales for annual N treatment rates ranging from 0 - 1344 kg N ha^{-1}. The highest recovery was achieved by perennial ryegrass; 63% of 252 kg N ha^{-1} was recovered, a proportion which decreased to 24% with increasing N application up to 1344 kg N ha^{-1}.

Denitrification losses approaching 8% and 4% of total applied N at Dunk and Brampton Island respectively were consistently maintained over all application rates and conditions.

By comparison, Mancino et al. (1988) found that denitrification losses from below turfgrass were strongly influenced by soil temperature. It was shown that 2 - 5% of applied N was lost from saturated silt loam soils at 22 °C and rose to 45 - 85% of applied N at 30 °C. Denitrification losses accounted for only 0.1 – 0.4% of the applied N when the soil was at 75% saturation. Velthof et al. (1997) also studied N_2O emissions from clay soil overlain by grassland and determined that total denitrification losses as a proportion of total N applied were large (8 - 14%) after wet seasonal conditions. They also concluded that denitrification increased proportionately with increasing N rate for grassland growing on a clay soil. Watson et al. (1992) measured total annual denitrification lost from grazed grassland to be up to 90% of the loss that occurred in the 0-5 cm depth. Ruz-Jerez et al. (1994) reported gaseous N losses form denitrification of the order of 1 - 1.3% of annual N input to fine sandy loam supporting white-clover cover.

As anticipated, the denitrification of N was negligible at Great Keppel Island due to its dryer soil profile and marginally lower soil temperature compared to the other study sites.

Also of interest were the temporal changes in the soil mineral N of each site. While there were pronounced shifts in the level of soil NO_3^- and NH_4^+ over monthly time periods observed in the output data, there was no significant overall increase in mineral N over 20 years for any island. This is a "common-sense" outcome – mineral N would only be expected to accumulate in soils under exceptional circumstances, therefore indicating confidence in the model.

The accumulated leaching envelope predicted for the four effluent irrigation regimes assumed for each island site are depicted in Figures 21 (a)-(c). This demonstrates that an increased usage rate of effluent on lawn areas promotes a similar increase in rate of loss of N from sites with a sandy geology like Great Keppel Island. The loss of NO_3^- through leaching ranged from 12.2% - 13.7% to 54.2 - 56.2% of applied N, an increase of 410% in applied N there.

By contrast, the envelope of variation of NO_3^- leaching from the soil profiles at Dunk and Brampton Islands were relatively uniform for the various hypothetical schemes of applied effluent N. Such limits fell between 10.6% and 14.6% at Dunk Island and 3.3% and 7.1% at Brampton Island. These results reflect the up to two orders of magnitude greater hydraulic conductivity of soils present at Great Keppel Island, as well as the lower proportion of N uptake by turfgrass predicted there. It should also be noted that surface water runoff was a major moisture loss mechanism at both Dunk and Brampton Islands – further reducing the available water for the transport of N.

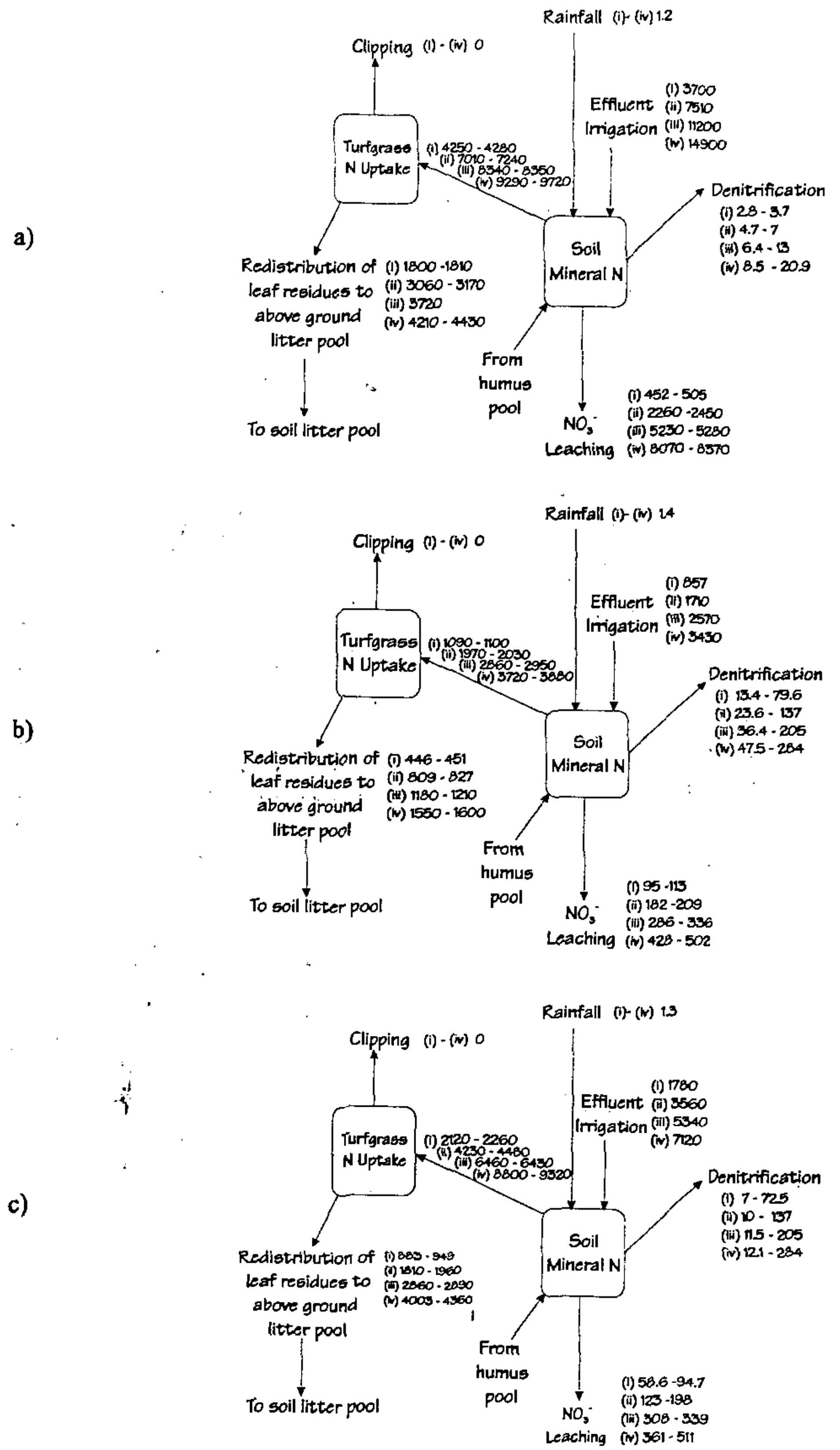

Figure 20. Major simulated N flows (kg N ha^{-1}) accumulated over 20 years at (a) Great Keppel, (b) Dunk and (c) Brampton resort islands for (i) 25, (ii) 50, (iii) 75 and (iv) 100% of total daily effluent irrigated.

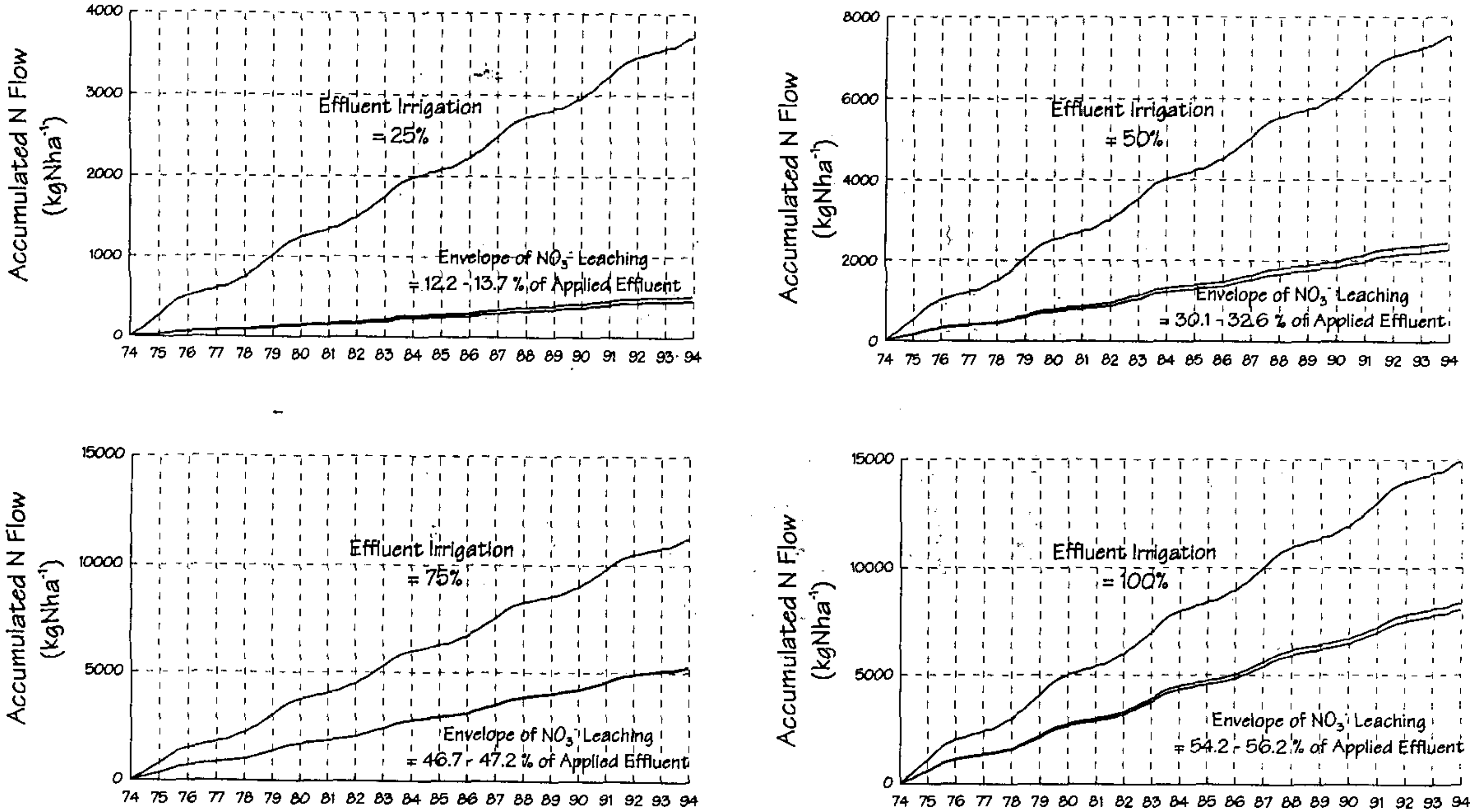

Figure 21(a) Accumulated NO_3^- leaching envelope during 1974 – 1994 at Great Keppel Island golf course for four effluent irrigation regimes.

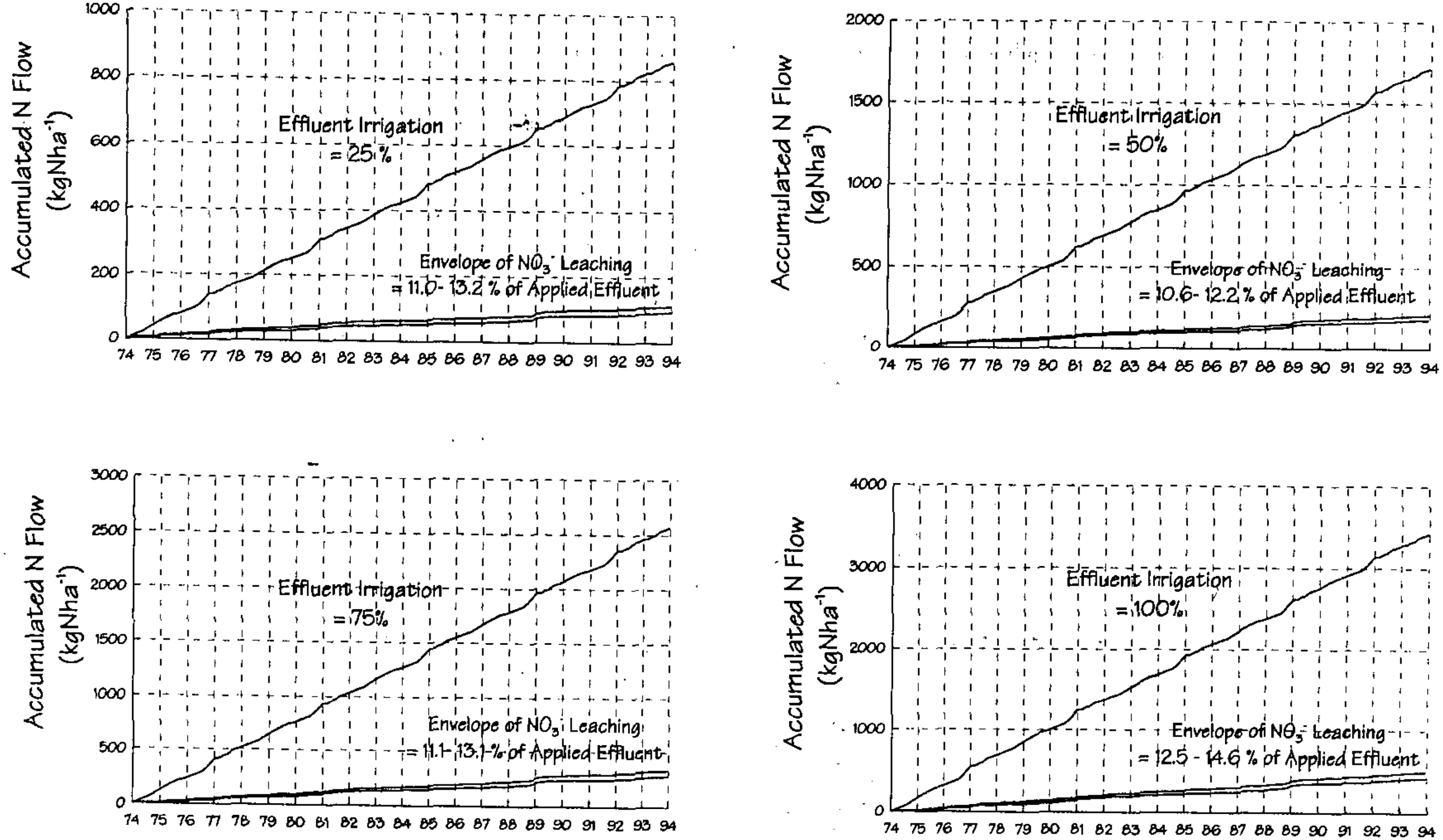

Figure 21(b) Accumulated NO_3^- leaching envelope during 1974 – 1994 at Dunk Island golf course for four effluent irrigation regimes.

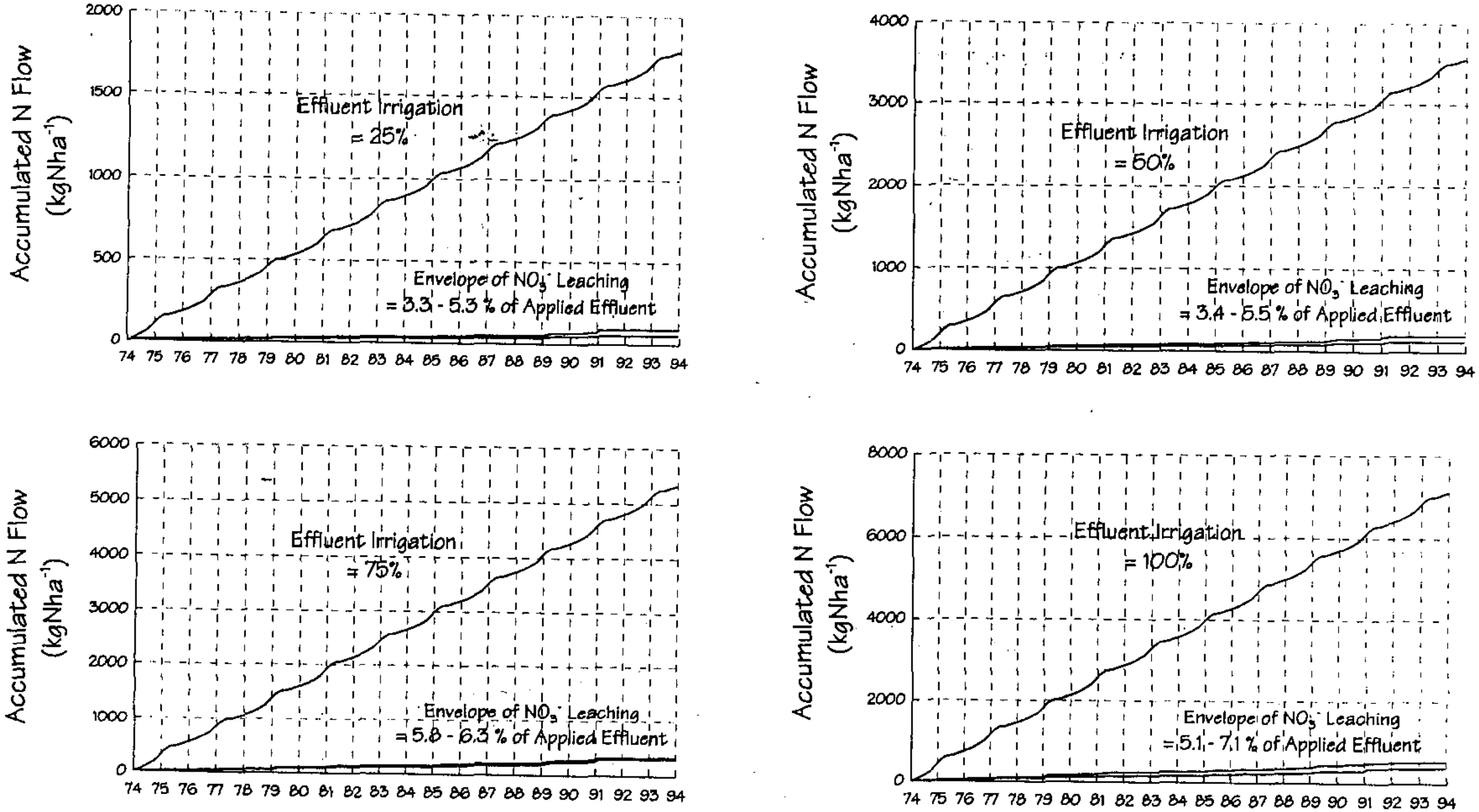

Figure 21(c) Accumulated NO_3^- leaching envelope during 1974 – 1994 at Brampton Island golf course for four effluent irrigation regimes.

Based on the golf-course areas of each resort, simulated mean annual losses of NO_3^- below the root zone at Great Keppel and Dunk Islands ranged from 30 to 502 kg N yr^{-1} and 28 to 126 kg N yr^{-1}, respectively, over all simulated wastewater application levels. Brampton Island currently employs effluent irrigation and thus, can expect to generate only between 7 and 38 kg N yr^{-1} flow beyond the root zone of the golf course (refer to Table 10).

Table 10. Comparative mean annual masses of NO_3^- output to GBR waters for various irrigation regimes over a 20-year simulation period for each GBR resort island.

Resort Island	Flow Path of NO_3^- to GBR Waters	Mean Annual Mass of NO_3^- (kg) Wastewater Applied as Irrigation (%)				
		0	25	50	75	100
Great Keppel Island	Below root zone	0	30	147	317	502
	Ocean outfall	894	671	447	224	0
Dunk Island	Below root zone	0	28	47	84	126
	Ocean outfall	858	643	428	214	0
Brampton Island	Below root zone	0	7	15	25	38
	Ocean outfall	534	401	267	134	0

The anticipated reduction in the N output to GBR waters from various effluent irrigation regimes when compared with complete wastewater discharge via ocean outfall were derived for each island site (see Figure 22). It should be emphasized that such estimates of the land-based output of N to sea were calculated on the basis that (i) no chemical or biological loss mechanisms existed for N solute in groundwater, (ii) there were no dilution effects and (iii) freshwater flow from the aquifer to the sea occurred in the vicinity of the site. In light of such conservative assumptions, the estimated export of N to sea would be expected to reflect the upper limit of expected outcomes. This would likely overwhelm any underestimation of N leaching caused by errors in the assumptions and parameterization of the numerical model.

At Dunk and Brampton Islands, the greatest reduction was achieved when all the daily effluent produced was distributed over an area equivalent in vegetation, soil characteristics and size to the existing golf course areas. In this case, an approximate reduction of 85% and 93% was simulated for each island respectively. With increasing fractions of wastewater applications, steady increases in the predicted mass reduction of N available to discharge to sea were evident.

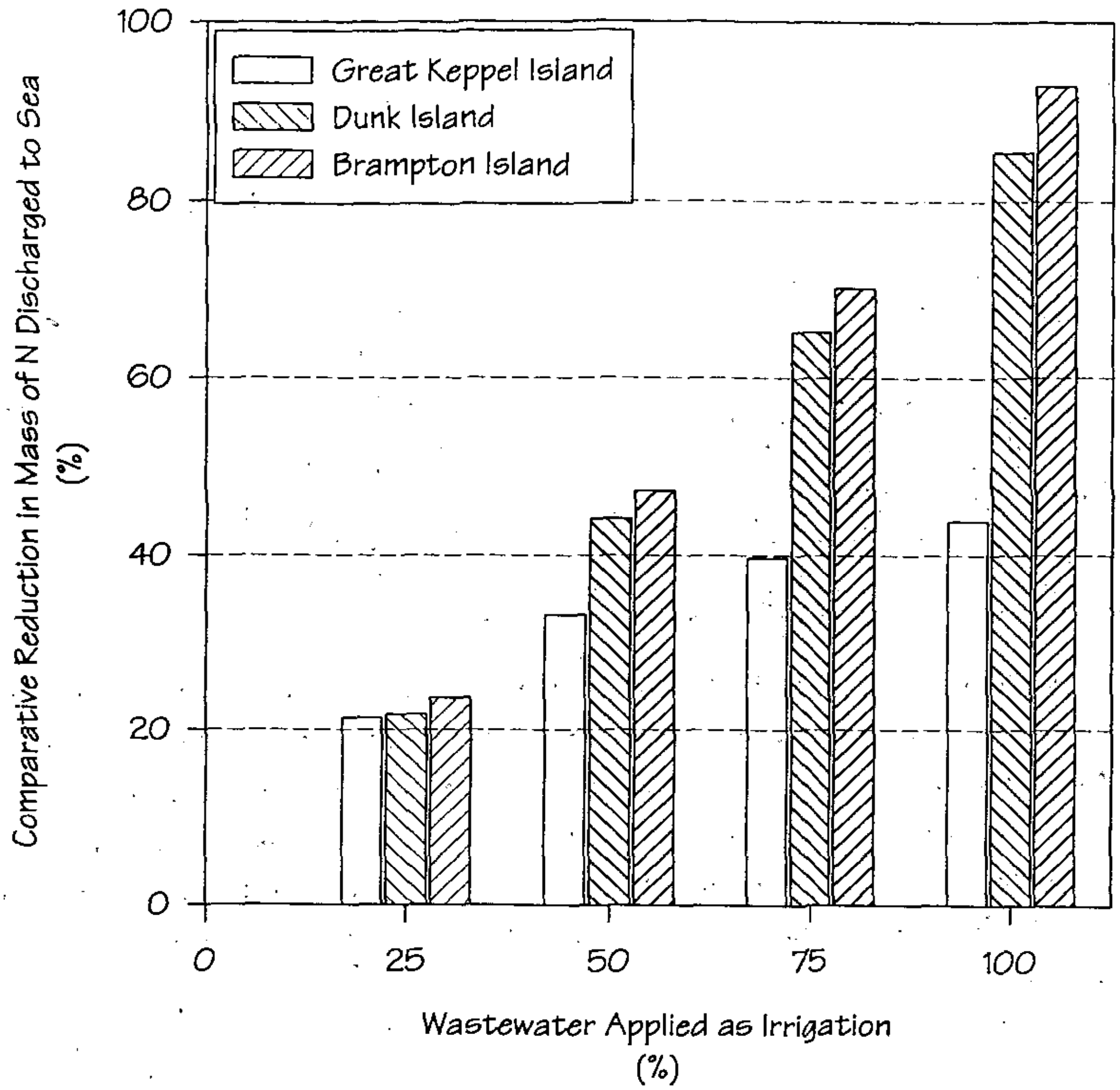

Figure 22. Reduction in NO_3^- mass outputs to GBR waters from effluent re-use at selected islands when compared with a complete discharge of sewage effluent via an outfall

Similarly for Great Keppel Island, the reduction in the flow of N beyond the root zone to potentially flow to GBR waters reached a maximum value for the peak level of wastewater irrigation loading. However, the reduction level of 44% was much less than at the other sites. Also, only relatively small increases in N reduction were predicted for substantial increases in the irrigation application rate. To put this in its proper context, the simulations undertaken for Great Keppel Island assumed an irrigation application area of 1.2 ha, thus larger areas could be expected to provide an improved degree of effectiveness in limiting N transport to the underlying aquifer.

To understand the role of the magnitude and frequency of rainfall on N leaching, the daily NO_3^- flow from below the lowest soil layer was examined for a one-year period at each site. Only the maximum effluent irrigation case was investigated, (see Figures 23 (a-c)). The peak outflows correlated with high rainfall events for all islands, particularly at Dunk and Brampton Islands while only limited leaching occurred during dry periods. As a consequence, pronounced flushing of stored NO_3^- from the lowest soil layer was expected to take place during high rainfall events. For Great Keppel Island, leaching was maintained during dry climatic cycles. This suggests a lesser influence of precipitation flushing events on the leaching fraction and more significant effects on the daily N outflows by consistently high-level effluent irrigation.

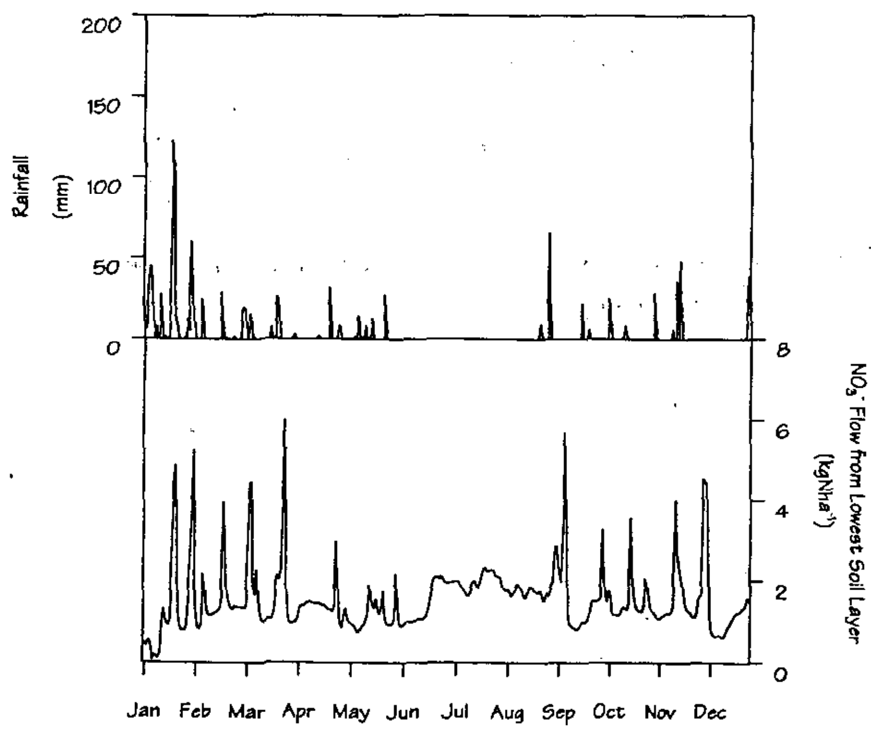

Figure 23(a) Simulated daily NO_3^- flows from below the Great Keppel Island soil profile related to daily rainfall inputs.

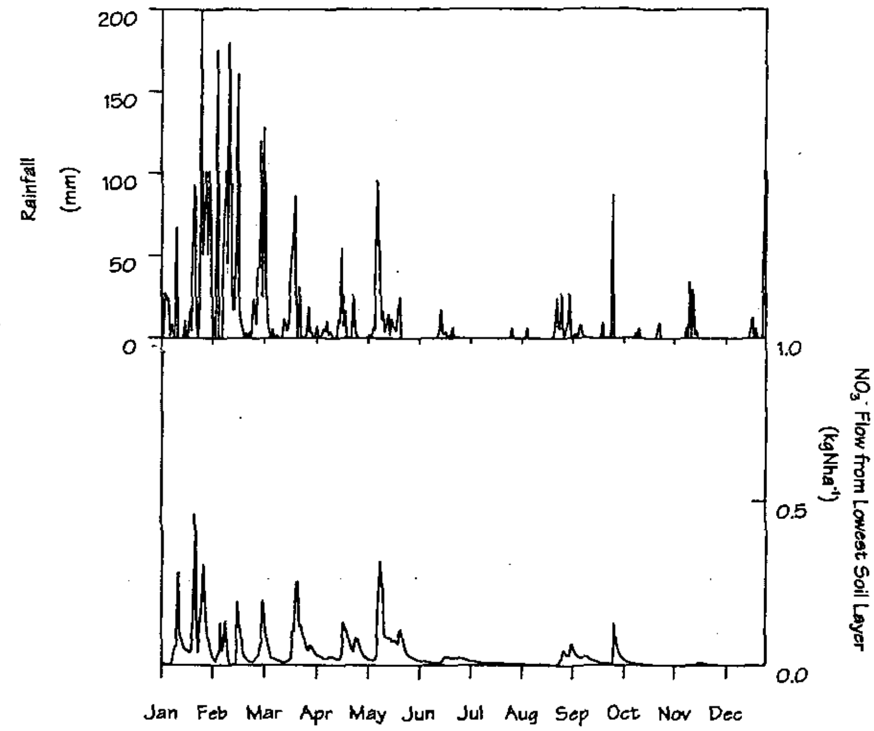

Figure 23(b) Simulated daily NO_3^- flows from below the Dunk Island soil profile related to daily rainfall inputs.

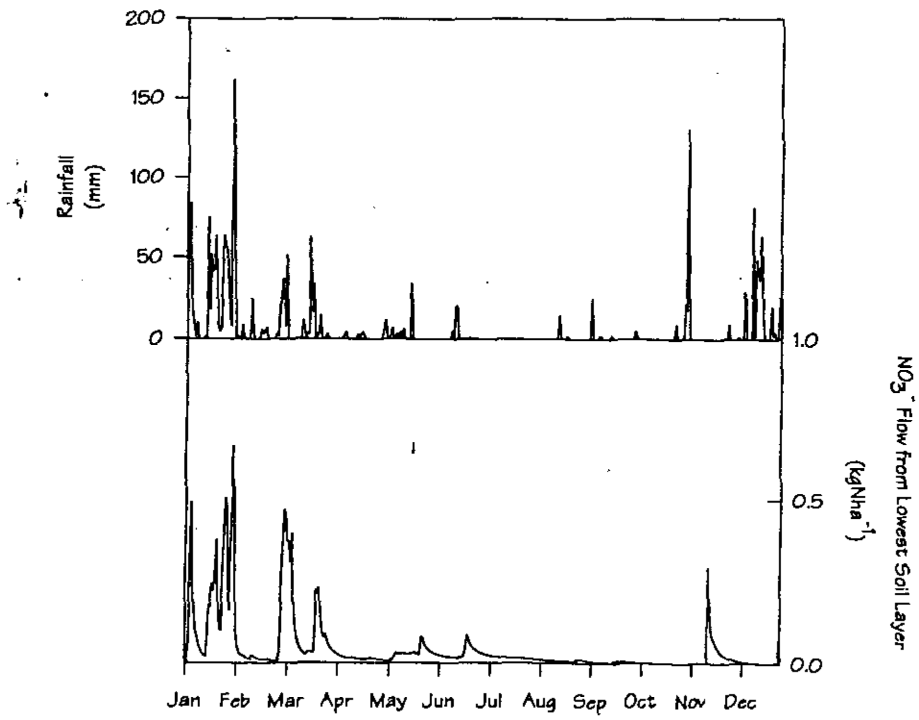

Figure 23(c) Simulated daily NO_3^- flows from below the Brampton Island soil profile related to daily rainfall inputs.

4.3 Sensitivity Analysis of SOILN Model

A sensitivity analysis is a study of the rate of change in one factor with respect to change in another factor (McCuen, 1973). Hence, the parametric sensitivity analysis of a model ascertains the impact of changes in one parameter on the outputs under review. To successfully apply this approach, a baseline data set was required to serve as a basis of comparison following the modification of each input parameter (refer Appendix 3). Since all deviations were interpreted proportionately to the baseline outputs, the exact specification of this baseline was not crucial.

Essentially, the purpose of this investigation was twofold. Firstly, to determine the *relative* influence of changes in SOILN parameter specification, and secondly, to ascertain the *quantitative* impact of parameter imprecision on the gross model outputs. In the process, the conservatism of the baseline or reference values adopted for this study could also be appraised.

One method for direct comparison of parameter influences on numerical model outputs is the quantification of the relative sensitivity coefficient (S_r) as described by Larocque and Banton (1994):

$$S_r = (\delta F/F)/(\delta X_i/X_i) \qquad (11)$$

where F is the baseline model result, X_i is the reference parameter value, and a variation in X_i (δX_i) results in a variation in F (δF). The value of S_r could vary from 0 (where the model result is unaffected) to >1 for an increasing effect of parameter change. If S_r =1, a specific percentage change in a parameter gave the same percentage increase in the model output. To strengthen the validity of this approach and simplify the interpretations made herewith, an increase in S_r caused by a small variation in the denominator of equation [11] was avoided by maintaining a constant ($\delta X_i/X_i$) across all parameters for each sensitivity threshold. The net result eschewed any bias from different widths of parameter interval.

Such a sensitivity analysis was systematically applied to 27 parameters considered the most likely to influence either the cumulative NO_3^- leaching (ACCDLOSS) or total plant uptake (ACCTOTUPT) for a 10-year SOILN simulation period spanning 1974 – 1983. By running the SOILN model with a series of individual parameter changes of up to ± 50 % from the baseline value, the effect of error in particular parameters was evaluated.

Only the Dunk Island site was considered in this report, for an effluent irrigation load of 2 mm d^{-1} or 50% of effluent produced daily, with mean soil hydraulic properties also adopted for this sensitivity analysis. To ensure that parameter variations were not symptomatic of unsteady state conditions, a 10-year simulation period was chosen to provide sufficient time for the N pools to equilibrate. In this new steady state, the annual mass balance of N was considered to remain constant.

A majority of the input parameters for the SOILN model were difficult to determine precisely, due to the complex experimental methodology required for their quantification as well as the interactions that exist between various N cycling processes. Model parameters that were able to be more routinely measured (soil N and C content) or were observable management practice (application rate and quality of effluent irrigation, grass clipping return rates, etc.) were not considered in this work. Importantly, SOIL model driving variables were not modified, implying that variations in physical and hydraulic characteristics of the soil (undertaken in Section 3.2), as well as meteorological factors, were not specifically accounted for during this sensitivity analysis.

Average relative sensitivity coefficients were determined for each of the selected parameters in terms of both the mean annual ACCTOTUPT and ACCDLOSS. The absolute values of each were used for comparative purposes and are shown in Figures 24 and 25.

For the ACCDLOSS results, with the exception of the UPMA parameter, all S_r were less than 1.0. Thus, for a specific change in the value of each tested parameter, a proportionally reduced change in accumulated NO_3^- leaching was output. As UPMA is an adjustment parameter for the fraction of mineral N available for immobilisation and plant uptake, it could not be measured in the field. However, it is highly unlikely that the baseline value adopted (UPMA = 0.08) was in error by more than ± 50% for these sites based on the range of values (0.05 - 0.12) reported in Jansson et al. (1991). Therefore, a rigorous calibration process and/or a critical literature evaluation of this parameter would be essential in future modelling work.

Higher S_r were associated with coefficients controlling photosynthetic growth through energy assimilation by turfgrass leaves (EXCOEFF, PHOEFF, NLEAFXG) as well as parameters defining N net mineralisation (LITTEFF, CPLANT).

Importantly, the EXCOEFF and PHOEFF were unlikely to deviate markedly from the assumed values, while the error in the estimation of NLEAFXG would be limited to no greater than approximately 20% (0.8% N) below the adopted value of 4 % N. In particular, the influence of these net mineralisation parameters on ACCDLOSS was anticipated, based on the large contributions to the total mineral N pools during net mineralisation of background humus and other organic matter.

The specific leaf area (WLAI) and the daily fraction of root biomass lost as litter (ALITTER(2)) were also found to be influential in the prediction of ACCDLOSS. The value selected for WLAI was not expected to be in error to the order investigated, as it was a directly measurable quantity that has also been well documented. Also, there is only a limited range over which the value of WLAI could be realistically applied.

Overall, several turfgrass growth and uptake parameters were clearly shown to have the greatest influence on the magnitude of ACCDLOSS. The continued growth of turfgrass in turn affects the overall distribution of available NO_3^- in the soil profile. These outcomes necessitated that close attention be paid to the accurate specification of these parameters. However, model predictions of NO_3^- leaching were shown to be relatively insensitive to the vast majority of soil N cycling parameters. Moreover, in the context of the endemic modelling assumptions and inherent difficulty in accurately measuring many input parameters, these results are considered an acceptable overall outcome.

In broad terms, soil moisture abiotic response factors (MOSM, MOS(1), MOS(2), MOSSA, MOSDEN) and denitrification (DENPOT, DENHS) were shown to produce lower S_r. It is important to note that SOILN employed these soil abiotic factors in non-linear functions to explicitly account for the effect on denitrification, nitrification and net mineralisation by moisture conditions. Consequently, the effect of these parameters would be expected to vary across each GBR resort island due to markedly differing soil hydraulic characteristics and precipitation. The sensitivity analysis demonstrated that these have only minimal influence on NO_3^- leaching and hence, require little or no specific quantification beyond reasonable limits for the SOILN applications used in this study.

For the sensitivity analysis of parameters relating to ACCTOTUPT, only four parameters were shown to be significant, namely: LITEFF, CPLANT, CNORG and LITHF. Of these, LITTEFF and CPLANT both yielded a $S_r > 1$. For LITEFF, with an effective parameter range of 0.2 – 0.7 (Alexander, 1961; Paul and Clark, 1988), extreme care was required in it's application by virtue of it's rather esoteric definition and the consequent difficulty in it's quantification. While CPLANT was almost as influential, the reference value applied was presumed to be sufficiently accurate due to its more restricted range of measure.

Table 11 gives the maximum percentage changes produced in the mean annual values, relative to the baseline set, for variations of up to ± 50% in major model parameters. While it was apparent that a greater number of parameters were highly sensitive for ACCDLOSS output, the percentage changes produced in ACCTOTUPT were more dramatic, with four parameters generating a change in ACCTOTUPT in excess of ±45%. Only one parameter for ACCDLOSS registered beyond ±36%. The maximum change in mean annual ACCDLOSS and ACCTOTUPT was an increase of approximately 115% for UPMA and 106% for LITEFF respectively for a parameter variation of ±50%.

A separate sensitivity analysis was undertaken of the effect on both ACCDLOSS and ACCTOTUPT caused by variations in the NO_3^- to NH_4^+ ratio (NITR) used in the nitrification function. This was due in part to the large range of possible values (1-20) quoted in literature (Jansson et al., 1991) coupled with a baseline value of 1 that would require > 50% variation to adequately monitor the effect of changes in NITR. When NITR was modified for a series of model runs, the maximum change in ACCDLOSS and ACCTOTUPT was 22.4% and 2.8% respectively for a change of up to +2000 % (or factor of 20). Therefore, the reference values would likely be adequate for setting this parameter. Notably, for scenarios where effluent irrigation was composed of high concentrations of NH_3, nitrification parameters such as NITR and NITK may have a greater impact on NO_3^- leaching because of a larger NH_4^+ pool, thus increasing the production of NO_3^-.

All other parameters had little or no significant influence in the simulated conditions, so an approximate determination of these parameters would appears sufficient. To test the reliability of the chosen reference values for these and all other parameters, a validation with field data was undertaken in Section 4.4.

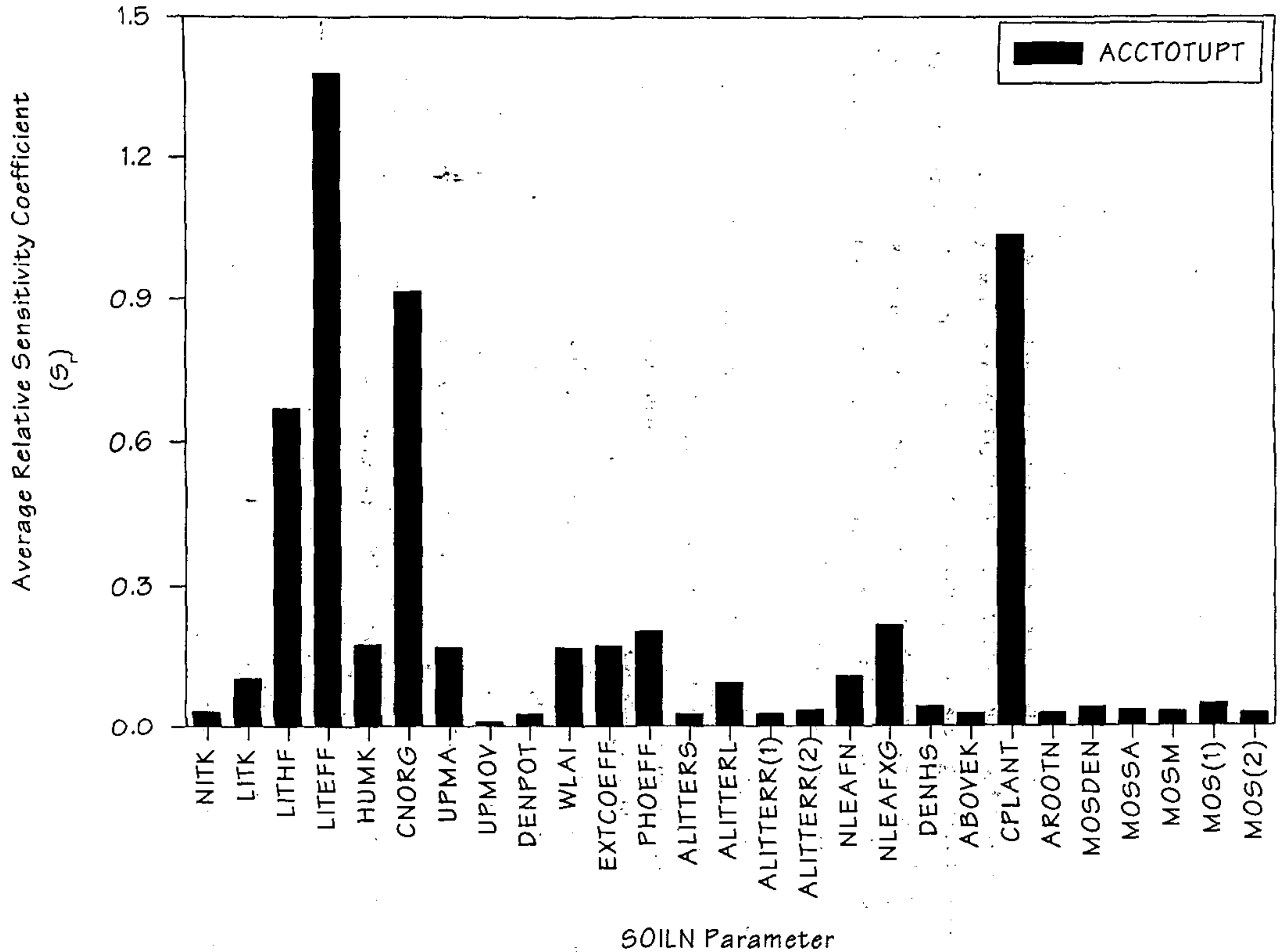

Figure 24. Average relative sensitivity coefficients (S_r) of SOILN parameters to the mean annual accumulated plant N uptake (ACCTOTUPT) at Dunk Island golf course with 50 % of sewage effluent irrigated daily over a 10-year simulation period. Note: S_r derived from a parameter variation of ± 50 %.

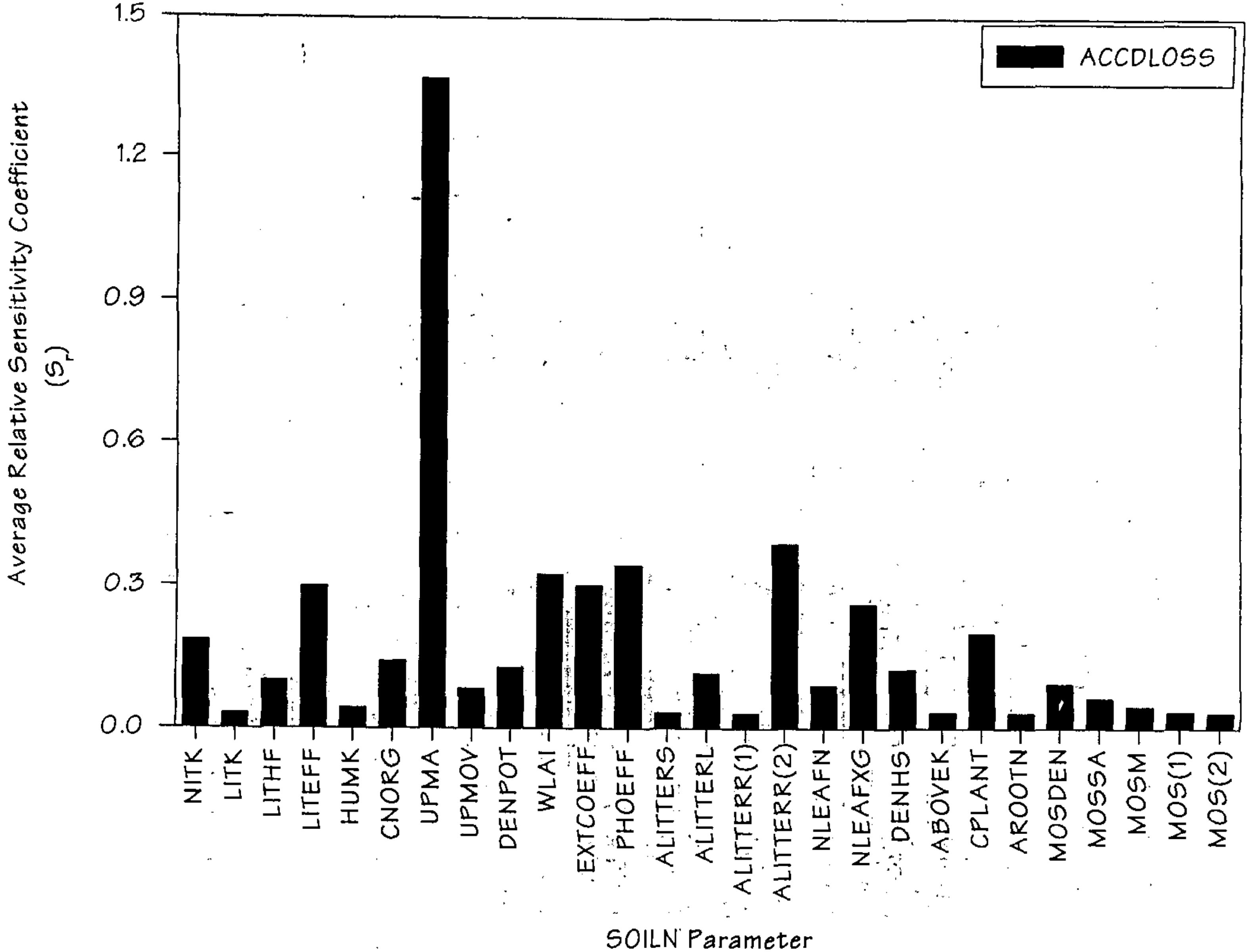

Figure 25. Average relative sensitivity coefficients (S_r) of SOILN parameters to the mean annual accumulated N leached (ACCDLOSS) at Dunk Island golf course with 50 % of sewage effluent irrigated daily over a 10-year simulation period. Note: S_r derived from a parameter variation of ± 50%.

Table 11. Sensitivity analysis of key SOILN parameters for Dunk Island golf course with 50 % of sewage effluent irrigated daily over a 10-year simulation period. The maximum percentage change in mean annual accumulated quantities, relative to the baseline set, are given for a parameter variation of ± 50 %.

SOILN Parameter	Max. δ in Mean Annual Accumulated Plant N-Uptake (%)	Max. δ in Mean Annual Accumulated N-Leached (%)
Baseline Total	*102 kg N ha^{-1} yr^{-1}*	*9.3 kg N ha^{-1} yr^{-1}*
NITR*	(-) 2.8	(+) 22.4
NITK	(+) 2.0	(-) 12.3
LITK	(-) 7.9	(+) 1.8
LITHF	(+) 47.1	(+) 10.0
LITEFF	(+) 105.9	(+) 30.1
HUMK	(-) 9.2	(+) 2.4
CNORG	(-) 45.7	(+) 11.6
UPMA	(-) 13.6	(+) 114.6
UPMOV	(-) 1.0	(+) 5.1
DENPOT	(+) 1.0	(+) 6.7
WLAI	(+) 11.8	(+) 31.3
EXTCOEFF	(+) 11.8	(+) 31.3
PHOEFF	(+) 13.7	(+) 35.8
ALITTERS	(-) 1.0	(-) 1.1
ALITTERL	(-) 6.1	(-) 6.5
ALITTERR(1)	(-) 1.0	(-) 1.7
ALITTERR(2)	(-) 2.5	(-) 22.6
NLEAFN	(-) 6.5	(-) 5.7
NLEAFXG	(-) 13.3	(+) 13.7
DENHS	(-) 2.9	(-)10.4
ABOVEK	(-) 1.0	(-) 1.3
CPLANT	(+) 79.4	(+) 19.9
AROOTN	(-) 1.0	(-) 1.1
MOSDEN	(-) 3.9	(-) 6.1
MOSSA	(-) 2.0	(-) 4.3
MOSM	(-) 2.3	(-) 3.2
MOS(1)	(-) 4.1	(-) 2.5
MOS(2)	(-) 1.0	(-) 1.7

(+) denotes that the maximum percentage change in mean annual accumulated quantities was an increase relative to the baseline value, with (-) denoting a decrease.

* denotes parameter variation of +2000%

When parameters react interdependently, systematic changes in individual parameters may not adequately represent field conditions. However, due to deficiencies in established knowledge concerning interactions between N cycling parameters, cross-correlation was not attempted. This was deemed an acceptable omission given the likely magnitude of experimental errors and the relatively small spatial scale being studied at each – possibly limiting the extent of heterogeneity in the parameter. This sensitivity analysis could be significantly enhanced in future work however, if applied to a wider range of climatic conditions, soil types and irrigation management practice. Nevertheless, the sensitivity analysis has provided an insight into the impact of parameter estimation errors in the outcomes from SOILN simulations for GBR resort islands, while identifying those parameters requiring particular attention during parameterization for other islands in the GBR region.

4.4 Validation of Predicted Nitrate Concentration Outputs from SOILN

An attempt was made to validate the predicted outcomes of N cycle modelling below turfgrass at GBR resort islands. This focused on the legitimacy of the simulated NO_3^- concentrations in the lowest soil cell being modelled. The golf course site at Great Keppel Island was chosen as a study site due to the detailed irrigation and fertiliser management data available for specifying external inputs to the model. Experimentally, it was considered to offer more accurate data by virtue of a less varied geology and greater suitability to the proposed sampling process to be discussed later. Of further interest was an examination of the leaching pattern over a shorter time-period at this site, where previously simulated leaching quantities far exceeded the other studied resorts.

4.4.1 Measurement of Soil Water Nitrate Concentrations

A network of 16 high-flow porous ceramic cups attached to 20 mm diameter UPVC tubing lengths were used to measure the temporal changes in soil water NO_3^- concentration at a depth of 1 m below the surface of the Great Keppel Island golf course (refer to Figure 26). The principle of operation was based on moisture transfer resulting from an induced pressure gradient. When suction was induced within a sampler cup, the capillary force of the soil at field capacity was opposed by the internal suction within the ceramic cup. When the soil solution suction was less than the applied vacuum, soil water was drawn across the porous wall into the cup. A removable self-sealing rubber septum atop each sampler served to hold the falling vacuum. This sampling process is illustrated in Figure 27.

Each cup was leached with dilute hydrochloric acid prior to installation to reduce contamination from fine ceramic particles remaining after production. After installation, the samplers were left undisturbed for six months before sampling commenced, to allow the surrounding soil conditions to equilibrate. Each network sampling routine was carried out shortly following irrigation and/or rainfall events to capture sufficient sample volumes representative of the downward migration of NO_3^-. As the adopted procedure was based upon a falling vacuum, the soil water was gradually collected within the sampler cup over 24 hours until collection – at which time they were immediately frozen and stored for later analysis.

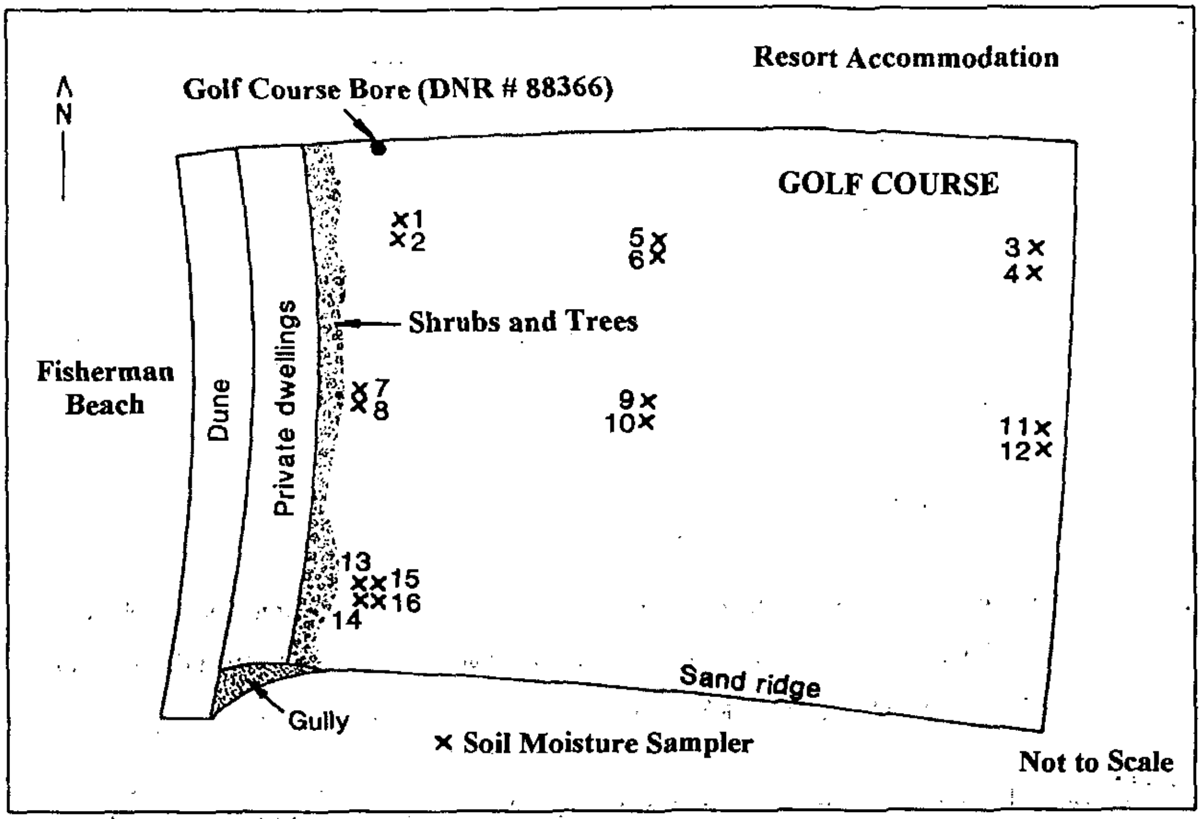

Figure 26. General location of the soil moisture sampling network at Great Keppel Island resort.

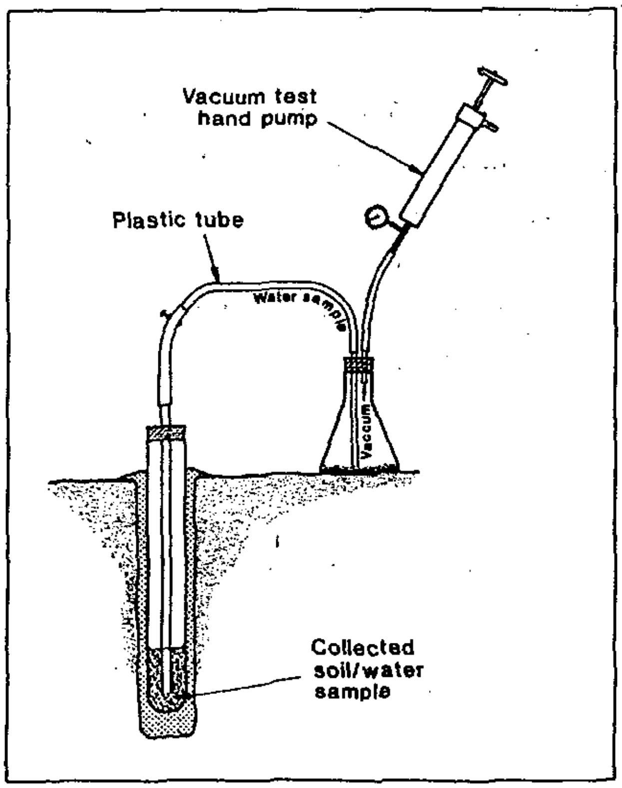

Figure 27. Generalized ceramic cup soil-moisture sampling procedure.

Soil Moisture Equipment Corporation (1995)

4.4.2 Applicability and Limitations of Ceramic Cup Soil Moisture Sampling

Numerous studies have explored in detail the inherent limitations of the field-scale assessment of nutrient concentrations using ceramic cup samplers. The overall validity of sample data obtained from this field study was therefore constrained by several factors.

Biggar and Nielsen (1976) asserted that soil solution samples could provide a good indication of relative changes in the amount of solute being transported, but not precise quantitative estimates unless the variability of the measurements was fully established. In an effort to reduce data variability, uniform sampler depths and sampling intervals in addition to the same initial applied vacuum were maintained for all sampling events at Great Keppel Island.

Van der Ploeg and Beese (1977) concluded that porous cups could draw water from a sphere of about 0.6 m and in so doing, could distort soil water flow patterns. They also concluded that soil moisture cups drain a wide variety of pores, not all of which contributed to the flow.

The potential maximum difference between the drainable soil water N concentration and the sampler concentration was shown by Hansen and Harris (1975) to equal the bias produced by departures between the sample intake curve and the soil water drainage rate curve summed with a 30 % variability they found arising from other sampling limitations.

The possibility of the porous cup samplers being circumvented by flow through macropores, instabilities or fingering phenomena should also be noted. However, this risk was thought to be minimal at Great Keppel due to the fine-textured uniformity of the sandy strata and the close contact achieved between the sampler tubing and the surrounding soil. Satisfactory results for N leaching studies were obtained by Barbee and Brown (1986) and Webster et al. (1993), and this approach appears valid for such soils. The coefficients of variation reported by Webster et al. (1993) were only 14-20% for example, and reflect the appropriateness of their use in sandy soil.

Another factor was the need for sufficient sample replications to achieve a representative value. Alberts et al (1977) showed that estimates of NO_3^- concentration in soil solutions taken from a silt-loam soil for six research plots of 16 m^2 would be within 5 % of the true mean if 246 replicates were carried out and within 30 % of the true mean for 10 replications. Due to the obvious logistical constraints and the low amount of sample volume available, such an exhaustive undertaking was not possible. Only two replications per sampler were carried out, but the moderately small study site size (1.2 ha), uniform vegetative and geological characteristics and consistent sample recovery approach were assumed to offset this concern sufficiently for the scope of this study.

4.4.3 Simulation Strategy

The simulation period for validation was set to the timeframe of measured results i.e. January 1996 to June 1997. Daily rainfall was measured onsite and other climatic data was again adopted from nearby Yeppoon. Sprinkler irrigation pumped from the adjacent golf course bore was carried out every three days for two hours giving a hydraulic loading of 2.5 mm d^{-1} per irrigation event. The sprinkler systems were assumed capable of distributing water evenly over the whole golf course area. Nutrient concentrations of water from the golf course bore were analyzed on three occasions during this period, averaging 3.1 mg NL^{-1} as NO_3^-, with negligible NH_4^+. Moreover, the N loading from irrigation was estimated as 0.07 kg N ha^{-1} applied as NO_3^- once every three days.

Slow release fertiliser was broadcast on fairways at an equivalent rate of 50 kg N ha^{-1} (80% as NH_4^+, 20% as NO_3^-) in mid-January and monthly to greens at a rate of 2 kg N ha^{-1}. Also, other than changes to the initial state variables - which were derived from the steady-state output of a 10-year simulation under these conditions, all SOIL and SOILN model parameters remained unchanged from previous Great Keppel Island simulations. The simulations were run for the two sets of soil parameters that previously yielded the envelope of probable deep percolation. The accumulated applied mineral N is depicted in Figure 28.

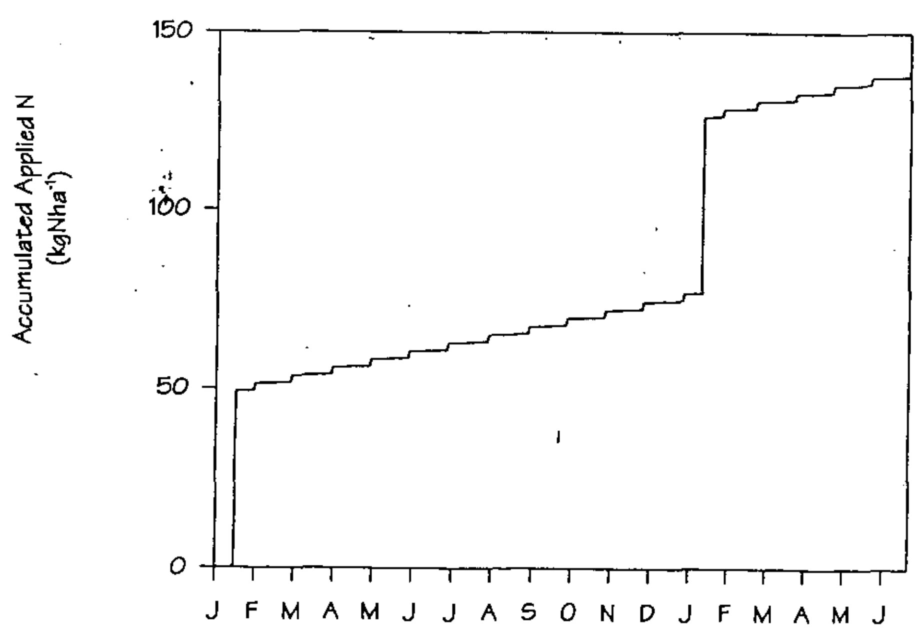

Figure 28. Total mineral N inputs from bore water irrigation and fertiliser at Great Keppel Island for the period Jan 1996 - June 1997.

4.4.4 Validation Results

Due to the errors introduced from the experimental limitations outlined in Section 4.4.2, the time-series of sampling data for soil water NO_3^- concentration were expected to serve merely as general indicators of the magnitude and pattern of the predicted outcomes from SOILN.

Even so, attention was still paid to statistical differences between measured and simulated results in view of the reasonable visual correlation of the measured data with the envelope of predicted NO_3^- concentrations in the lowest cell (refer Figure 29).

The measured data were skewed spatially and followed a log-normal distribution of values. This was due in part to the uneven fertiliser coverage and the influence of topography. A geometric mean (equivalent to the mean of the log-normal distribution) was taken over the extent of the site to reduce the impact of any inordinately large values as recommended by Addiscott (1994).

Predicted concentrations of NO_3^- followed annual patterns expected in sub-tropical grassland environments. During the winter period, when uptake of N was relatively low, NO_3^- leaching was highly dependent on the timing of fertiliser application and the degree of soil water percolation at depth. During the summer months, when temperatures were high and water conditions near optimum for turfgrass growth, utilisation of NO_3^- was more efficient due to the corresponding increases in grass photosynthetic rates and higher denitrification during wetter soil conditions.

The spike in predicted NO_3^- concentration that occurred in early May 1996 was influenced by a succession of high rainfall events in late April spanning 10 days. This stimulated a rapid movement of residual NO_3^- through the highly permeable soil, but unfortunately the magnitude could not be fully validated as sample capture occurred just after the simulated peak.

The apparent overestimation of concentrations immediately following the broadcast of fertiliser in mid-January 1997 could be a reflection of differences in the actual and assumed dissolution rate of applied fertiliser. Fertiliser additions in the model were assumed to equilibrate quite rapidly with the NO_3^- in the top layer. In reality, applied N may have remained in the uppermost soil layer for an indefinite lag time, generating N losses by NH_4^+ volatilization that were not accounted for in the SOILN model.

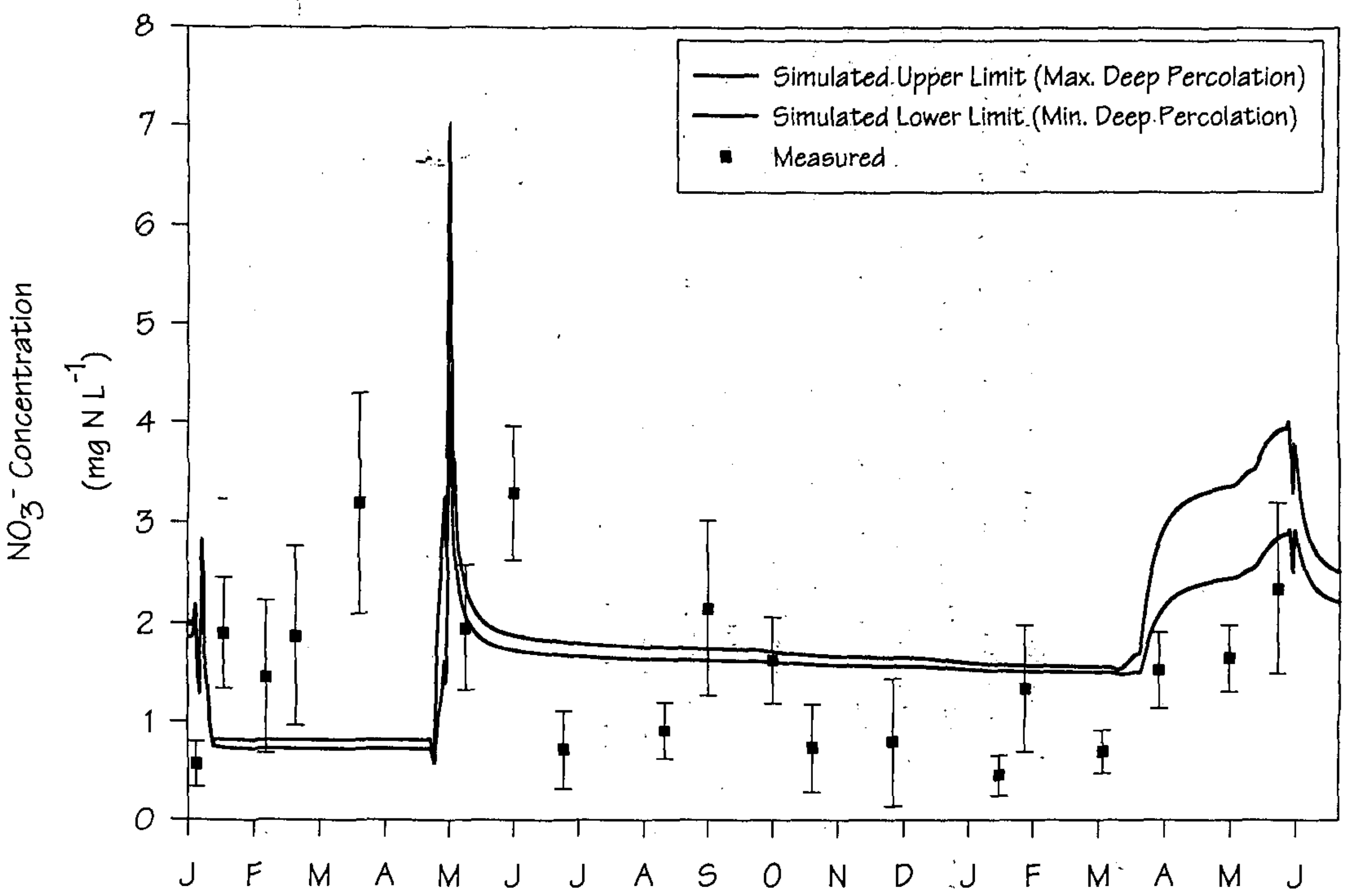

Figure 29. Measured values and model prediction envelope for a depth of 0.85 - 1.0 m below surface level at Great Keppel Island for the period Jan 1996 - June 1997. Measured concentrations are geometric means. Error bars denote plus and minus one standard deviation.

Ammonia volatilization losses have been estimated at 4-30% in studies of NH_4^+ rich additions to pasture soil in temperate areas (Sherlock and Goh, 1984), and would have been exacerbated by the hot, dry conditions that prevailed during this period at Great Keppel Island. Consequently, less than predicted concentrations of NO_3^- measured in soil solution at 1 m depth were not surprising. Conversely, the higher than predicted NO_3^- concentrations measured in the lowest soil cell following the initial fertiliser application in mid-January 1996, were difficult to rationalise. The dynamic simulation of NO_3^- may have been improved if drainage flow could be simulated more precisely. An explicit validation of predicted soil water flux has yet to be attempted.

Soil sampling for the determination of NO_3^- in the profile was conducted on three separate occasions during the validation study: following each annual fertiliser broadcast (day number =70, 476) and approaching the end of the annual fertiliser cycle (day number = 320). For each sampling event, soil cores were obtained from four representative locations within the golf course and sectioned into the same discrete partitions adopted in the model. These were then bulked into one composite sample per depth increment. Soil samples were deep-frozen on the day of collection to inhibit conversion of N. Once thawed, 100 g of the moist soil was extracted with 250 mL 2 *M* KCL. Nitrate concentrations were determined in the extracts using automatic colorimetric analytical methods in accordance with Australian Standards. The values were converted into kg N ha^{-1} using actual water content from the gravimetric method and dry bulk density values determined for each soil depth horizon.

A comparison of the simulated and observed depth profiles for soil NO_3^- content at day number 70, 320 and 476 is given in Figure 30. In general, there was reasonably good agreement in relation to the approximate magnitude and distribution of soil NO_3^- within the profile for the individual sampling events.

The measured distribution of NO_3^- at day number 70, which was preceded by a major annual broadcast of fertiliser at day number 17, was found to have shifted deeper than predicted. This may have been due to an underestimation of the initial conditions for mineral N. At day number 320, the simulated build-up of NO_3^- below approximately 0.4 m that had taken place since day number 70 demonstrated that sufficient rainfall and/or irrigation had transferred NO_3^- beyond the simulated soil profile or had been lost to denitrification. Again, measurements supported this trend, although, there was a noticeable lag in the downward migration of soil NO_3^- in the measured data. By day number 476, following the most recent major application of N occurring at day number 384, an increased accumulation of NO_3^- in the soil horizons below 0.6 m was predicted and validated by measurement.

a)
b)
c)

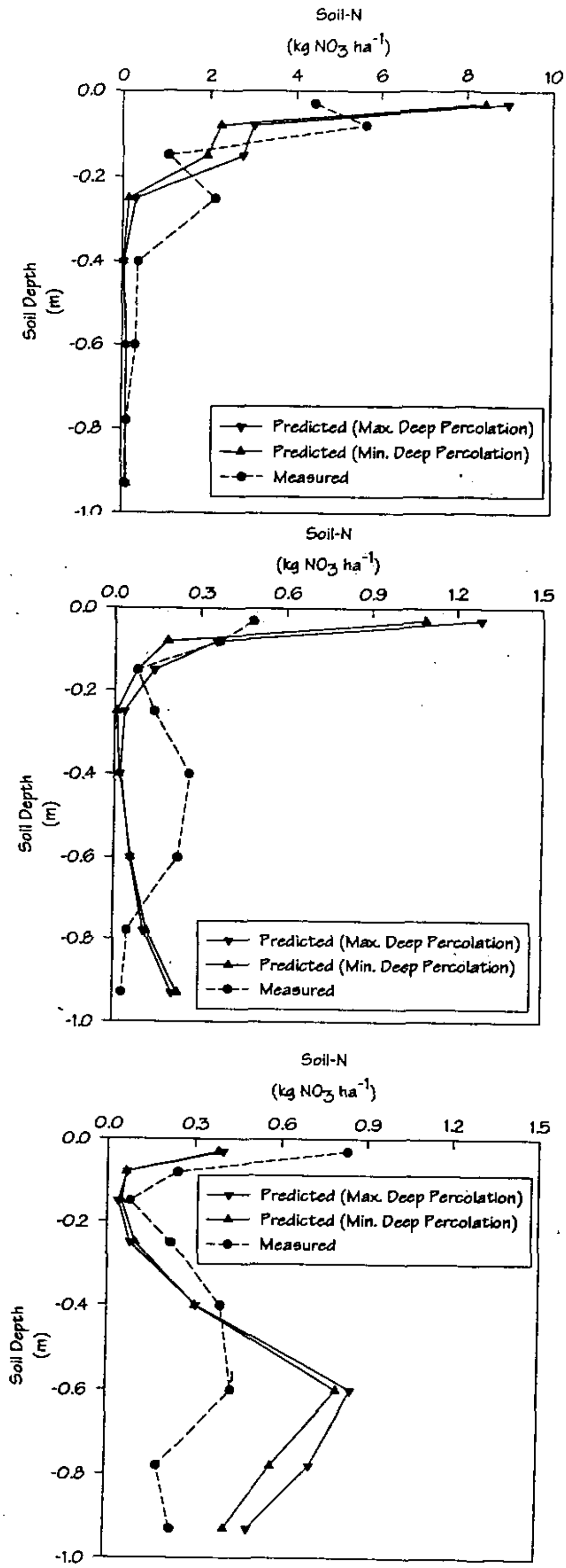

Figure 30. Comparison of simulated and measured depth profiles for soil NO_3^- (kg N ha^{-1}) at: (a) day number = 70; (b) day number = 320; (c) day number = 476.

Evidently, the absence of high water flux in the lower profile had prevented significant migration of soil NO_3^- beyond the effective root zone. Once again, the observations lagged the simulated output for downward soil NO_3^- distribution with depth, which lends support to the application of this specifically configured SOILN model as a tool for generating *conservative* estimates of NO_3^- leaching losses for turfgrass areas at GBR resort islands.

Given the anticipated spatial variability of soil NO_3^- combined with experimental error, the degree of discrepancy between simulated and measured soil NO_3^- was considered acceptable and did not invalidate the broad-based outcomes and trends illustrated by the SOILN model.

4.4.5 Statistical Data Analysis

To more accurately assess the reliability of the SOILN model simulations, the coefficient of determination (R^2) was presented as a measure of the association between observed and predicted NO_3^- concentrations at a depth of 0.85-1.0 m below the surface. Two other statistical properties were also calculated to evaluate the differences between predicted and measured values. The mean residual error (ME) was defined as:

$$ME = \left(\frac{1}{n}\right)\sum_{i=1}^{n}(x_i - y_i) \qquad (12)$$

and the mean squared residual error (MSE) was defined as:

$$MSE = \left(\frac{1}{n}\right)\sum_{i=1}^{n}(x_i - y_i)^2 \qquad (13)$$

where n = number of data points of predicted and measured NO_3^- concentration, x = measured NO_3^- concentration and y = simulated NO_3^- concentration.

Mean residual error is a measure for the bias in the simulation output. Values approaching zero indicate that measured and calculated values do not differ systematically from each other, or equivalently, that there is no consistent bias.

Mean squared residual error is a measure of the scatter of the data points around the 1: 1 line. High MSE values indicated large scatter and also imply high ME values. Conversely, low MSE can serve to establish a high level of correlation.

The ME ranged between -0.29 and -0.06 mg N L^{-1} for the effective range of deep percolation simulated by SOIL. The negative ME values indicated that measured NO_3^- concentrations were systematically lower than the simulated values. This is supported by the regression of measured versus predicted NO_3^- concentrations (Figure 31), where the majority of data points are found below the 1:1 line. The MSE ranged between 1.37 and 1.05 (mg N $L^{-1})^2$ across the range of predicted deep percolation modelled. These are quite high as expected from the rather large scatter around the 1:1 line. Figure 31 provides a visual reflection of the statistical analysis in terms of bias and scatter, with the slope of the regression lines (R^2) varying from 0.71 to 0.84.

Systematic deviations as expressed by the ME values, could have been corrected by model calibration. Although, it was probable the systematic overestimation of NO_3^- concentration by the model was due more to shortcomings in the recovery strategy, in terms of the deficiencies in the ceramic cup sampling technique and the limited number of measured points in time and space. In view of the tendency of the model simulations of NO_3^- concentrations in the lowest cell to be reasonably conservative, the modelling parameters were considered satisfactory in predicting upper limits for leaching.

Similar magnitudes of error were found in a number of previous modelling investigations for NO_3^- leaching below vegetation sustained on sandy strata, (Lewan, 1994; Blomback et al., 1995). Lewan (1994) found that ME values from simulated and measured NO_3^- in deep drainage varied from -1.72 to $+1.11$ when 90 kg N ha^{-1} yr^{-1} was applied to sandy-loam supporting cereal crops in southern Sweden. Blomback (1995) reported R^2 values for measured versus predicted soil NO_3^- content that ranged from 0.9 (n=18) to as low as 0.38 (n=18) for N inputs of 230 kg N ha^{-1} yr^{-1} to a sandy loam growing wheat in Germany.

It is important to note that the deviations between the observed and predicted data sets of this study were derived from a much larger experimental unit (and hence with higher probability of error propagation) than those in Lewan (1994) and Blomback et al. (1995). Overall, given the various constraints discussed, the agreement is considered quite good.

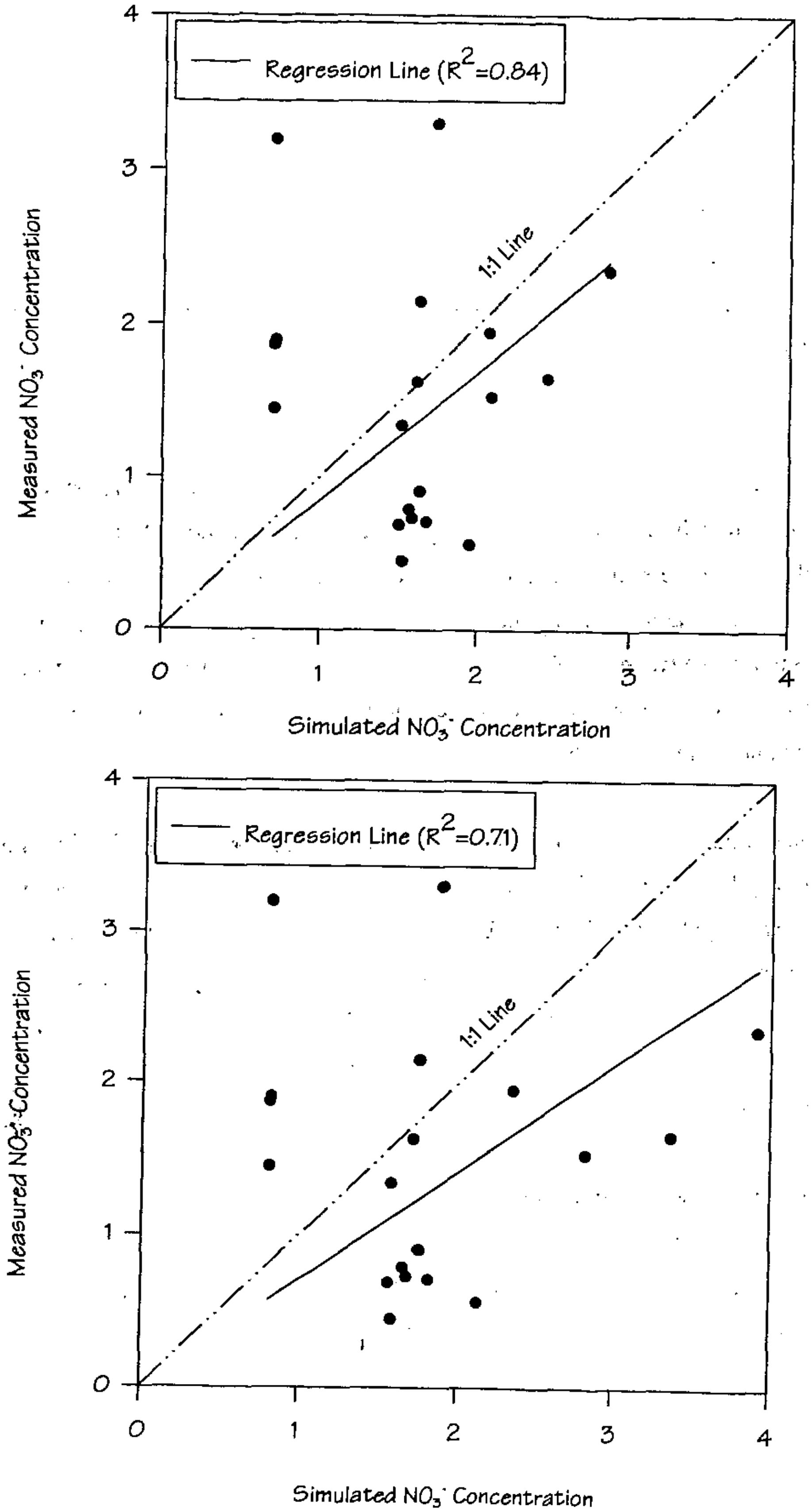

Figure 31. Comparison between simulated and measured NO_3^- concentrations (mg L^{-1}) at Great Keppel Island for an 18-month period from Jan 1996 to June 1997.

5. DISCUSSION

5.1 Performance and Utility of the Model

Simulation modelling allows the exploration of the complex system in question, indicating directions for future research with the ultimate aim of better understanding of that system (Kirkby et al., 1993). An important outcome of this approach is the identification of inadequacies and limitations in empirical work and theoretical assumptions, as well as an appreciation for the governing processes.

A large number of parameters needed to be specified in order to run the SOILN model. Many of these were only considered appropriate for specific applications, since they vary according to site or other conditions. In view of this, it may be difficult sometimes to apply the model astutely for a particular scenario due to a lack of knowledge regarding representative parameter values or the unavailability of suitable experimental data.

The sensitivity analysis described in Section 4.3 was able to illustrate the factors most likely to influence the predicted N export from below the root zone. An approximation of the sign and magnitude of model output changes was given for key parameters.

The value of the parameterised model as a predictive tool and scenario tester was also demonstrated for soil NO_3^- leaching from nutrient-rich effluent irrigation. A very high proportion of applied N was recycled as cut grass biomass, owing to the perennial growth and nutrient uptake of grass, which ultimately, exacerbated predicted N losses from the root zone. All but the higher effluent irrigation rates at Great Keppel Island predicted a proportion of NO_3^- transported below turfgrass root systems that was small in relation to plant uptake when considered over a long time scale. Large percentage errors in the amount of leached NO_3^- were actually small errors in relation to the high annual flows in the system, which inevitably occurred unless the representation of turfgrass growth was precise. Such accuracy was difficult to achieve when modelling biological processes.

Discrepancies between the measured and predicted values of soil and leachate NO_3^- at the Great Keppel Island site could have been caused by a combination of experimental and model formulation error. The precision with which soil N and C cycling was represented by SOILN depended on an accurate description of the main constituent processes, which at present, omits ammonia volatilization and is based on a host of simplifying assumptions. An accurate description of NO_3^- leaching also depends on the description of water movement in the SOIL model, especially the deep percolation rate to groundwater and surface runoff volume. The likely sources of experimental error were discussed in Section 4.4.2.

Despite these errors, the model appears to provide a realistic representation of the relevant processes in broad terms. For N cycling studies, it gives a good indication of the relative quantities of nutrients associated with each route through the system and a meaningful upper limit to the availability of N to flow to sea.

No attempt was made to predict any set of experimental values by exactly synchronizing or matching simulation and experimental outcomes for the N concentration in leachate, due to spatial variability issues and the exhaustive data required for such an exercise. This research has instead been guided by a literature-based understanding of these systems, in conjunction with an 'order of magnitude' validation process. Future model refinement for field scale applications at GBR resort island sites may best be directed towards dry-matter production of grass clippings, which are a more routinely measured system output.

There is undeniably much room for improvement in the fit obtained between SOILN model output and measured observations in relation to NO_3^- concentration below the root zone for the GBR resort island applications investigated. This would certainly require further enhancement of the concepts and methodology upon which the model is founded, while an improved estimation of model parameters and field measurement technique appears necessary. Further investigation is also required into the potential of the model to effectively quantify N leaching from a more diverse range of effluent irrigation and fertiliser management strategies than those presented in this study.

While uncertainties exist for many fundamental properties that control N flows in soil, if a sound balance of independent field data and model outcomes can be achieved, improve the quantitative assessment of effluent irrigation practice at GBR resort islands. It is hoped that even in the absence of detailed measurements of model parameters, as is generally the case, the SOILN model (as configured for this research) would be sufficiently robust, and transportable in terms of turfgrass parameters. This issue will be addressed more comprehensively in a future management-oriented report focusing on a wider range of effluent irrigation, soil type and meteorological permutations at GBR resort islands.

5.2 Outcomes for Management

Logistical considerations involving the implementation of effluent irrigation on GBR islands include land application area requirements, water storage pond sizing, public health and aesthetics. Indeed, large tracts of land may be required for effluent irrigation.

Hewitt (1990) estimated that for a hypothetical 1000 EP plant, an area of around 20 hectares would be required to ensure no runoff or direct discharge of effluent in up to the design 1 in 10 year wet year. He further intimated that a wet weather storage pond providing 90-120 days capacity would also be required to balance out dry weather and wet weather irrigation application rates. While the peak occupancy rates at GBR resort islands are often much lower (refer Table 1), substantial land areas for effluent irrigation are still likely to be prescribed using traditional methods such as the New South Wales EPA model (EPA, 1995) and Victorian EPA model (Thomas, 1992).

It is therefore intended that the SOILN model configured for this report be considered as a planning tool for the optimisation of land areas and effluent irrigation scheduling. For given effluent production rates and N quality, simulations could be run to ensure minimal adverse effects on aquifer water quality and thereby restrict long-term rises in groundwater outflow of N to GBR waters.

Based on the simulation results undertaken to date, daily applications on the golf course areas at Brampton and Dunk Islands would appear to be acceptable (in terms of N migration) for any proportion of effluent currently produced.

However, for heavily irrigated effluent on a predominantly sandy island such as Great Keppel Island - which features a water table close to the surface, the highly mobile N was predicted to readily leach through the unsaturated zone into the aquifer. As this aquifer is also used for resort water supply, the utility of this resource could be impaired over time. Therefore, any future effluent irrigation proposal at Great Keppel Island would need to include the airstrip and gardens for example, given that all other considerations are met. This would serve to reduce hydraulic loads and possibly disperse the source sufficiently to enter other hydraulically disconnected aquifer systems.

As turfgrass growth proceeds all year round, NO_3^- can be absorbed from the soil almost continuously. In some cases, proper timing of residue incorporation could decrease the potential for NO_3^- leaching by immobilising some of the residual N that otherwise might leach. Although, the subsequent mineralisation that occurs during biodegradation may function as a slow-release form of N fertilisation and could unduly promote the release of N into the aquifer system when coupled with effluent irrigation inputs. It is therefore essential that the turfgrass be clipped regularly and removed off the irrigation area for correct disposal. Composting clippings with dried sludge from the sewage treatment process is one option.

Some further recommendations regarding the practical implementation of an effluent reuse scheme at a GBR resort island are as follows:

- Development of a wet-weather area at a satisfactory distance from the nearest potable bore is imperative to minimise the impact of flushing effects on N distributed lower in the soil profile from excessive and/or prolonged rainfall - as demonstrated for Dunk Island;
- A soil depth of at least 1.0 m is considered adequate, since this was shown to be sufficiently deep to allow regular root development and provide ample residence time to generate N losses and transformations in the biologically active soil zone;
- Benefits of irrigation at night are threefold: (i) wind velocities are generally lower, thus reducing the distortion of irrigation patterns by minimising windblown spray; (ii) limit public exposure to aerosols thereby alleviating public health concerns and (iii) lower evapotranspiration loss produces greater efficiencies of water use and therefore satisfying turfgrass moisture requirements for optimum growth;
- Windbreakers and/or buffer zones should be allocated around the site.

- Fuller and Warrick (1985) suggest that wastewater application should be limited to lands with slopes less than 6% to avoid excess surface water runoff losses to nearby watercourses or directly to sea and limit particulate nutrient transport by erosion;
- To avoid lower consumptive use by turfgrass that occurs when a non-uniform distribution of irrigation exists, astute spacing and selection of sprinkler heads is required to optimise coverage. For example, subsurface or pop-up sprinklers with low-pressure nozzles may be appropriate, or in mechanically moved systems, the lateral could be fitted with spray nozzles directed downward and applying the effluent close to the ground.

This research will culminate with future general guidelines for effluent disposal practices on GBR resort islands. It is hoped that they may provide the framework and necessary background data for use in updating policy for GBR effluent irrigation management in relation to N exports to sea. This document will provide recommendations regarding maximum advisable effluent application rates, allowable limits of effluent N concentration and scheduling considerations, with reference to a variety of island geology and seasonal conditions.

6. CONCLUSIONS

Irrigation re-use of effluent discharged from coastal resorts provides an alternative to the discharge of treated sewage direct to surrounding waters and also a possible opportunity for effective management of water resources. In effect, it recognizes the ability of land to reduce and retard the movement of N into the adjoining aquatic ecosystem.

To evaluate the advantages and risks of sustained effluent irrigation, a soil water flow model and N transport sub-model were employed to predict the soil N balance for a 1 metre deep soil profile for three island resorts over a 20-year period. The predictions were based on hypothetical schemes of effluent irrigation deduced from the available effluent data for each resort.

The assumed application areas at Dunk and Brampton Islands were shown to be sufficient to ensure an adequate reduction in the transfer of N from the vadose zone for all cases considered. Notably however, the high annual rainfall at Dunk Island could increase the likelihood of rapid transport of N via surface runoff and short circuit flow paths.

For low levels of applied effluent N to Great Keppel Island, the turfgrass and soil profile were reasonably effective in limiting the transfer of N to the aquifer system. However, at medium to high rates of applied N the soil characteristics lead to a more pronounced degree of N migration. For irrigating large fractions of the total available sewage effluent, it would be prudent to distribute over a greater area than that assumed in this study to reduce the hydraulic loading.

It also should be noted that significant N contributions to each island system from the non-removal of grass clippings were predicted. This counteracted somewhat the high degree of effectiveness of turfgrass in utilizing applied N, so that clipping removal is recommended to further diminish potential leaching risks from moderate to high effluent reuse applications.

Based on the level of leaching predicted for Great Keppel Island at high levels of applied N, elevation of the nitrate concentration of surrounding waters, sufficient to be detrimental to the marine ecology, cannot be discounted. Further detailed studies of resort island nutrient mass balances are warranted in conjunction with the need for more detailed work on the groundwater dynamics and the residence times around the coast before transport away by currents and tides.

For both Dunk and Brampton Islands, the maximum reduction in the potential N flow to GBR waters was achieved when 100 % of the current daily effluent production was distributed over a turfgrass area corresponding to the size of each golf course area. A reduction of 85 % and 93 % was simulated for each island respectively in such a case. As wastewater application rates rose, steady increases in the simulated reduction of N available for ocean discharge were predicted. This suggested that if such high levels of effluent irrigation usage are viable from an economic and logistical standpoint (subject to public health concern) then the heightened degree of effectiveness in reducing potential discharge quantities could be capitalized on.

The reduction in the flow of N below the root zone at Great Keppel Island also reached a peak value for the maximum level of wastewater irrigation loading. The degree of effectiveness of land utilisation of N was not as high as at the other islands studied, however, the potential reduction in the mass of N exported to GBR waters reached 44 %. Additionally, relatively small increases in N usage by the land system were associated with much larger increases in the irrigation rate, pointing to a maximum threshold of N uptake by turfgrass being approached or surpassed. For this study, the prospective irrigation areas of each island site were assumed to be no greater in extent than the golf course, whereas larger areas within the resort environs such as airstrips and gardens are likely to be available for use. As a consequence, the simulations undertaken should be considered as a lower estimate of N reduction in terms of the level of N available for discharge to the local marine environment. This is further supported by the conservative assumption of N as a non-interactive solute entering an aquifer (with no dilution), which freely discharges to the sea.

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9. Appendices

Appendix 1a. Great Keppel Island golf course measured hydraulic properties – statistical data

Soil Layer *[cm]*	*σ Porosity* θ_s *[%]*	*S.D Porosity* θ_s *[%]*	*CV Porosity* θ_s *[%]*	*σ Wilting Point* θ_{wilt} *[%]*	*S.D Wilting Point* θ_{wilt} *[%]*	*CV Wilting Point* θ_{wilt} *[%]*	*σ Saturated Conductivity* K_{sat} *[cm* d^{-1}*]*	*S.D Saturated Conductivity* K_{sat} *[cm* d^{-1}*]*	*CV Saturated Conductivity* K_{sat} *[%]*
0 – 5	47	3.7	7.8	10.2	2.4	23.8	62.0	31.1	50.2
5 - 10	39	2.3	5.9	8.7	2.7	31.1	47.0	21.2	45.1
10 - 20	43	1.0	2.3	5.0	0.3	6.2	310	57.7	18.6
20 - 30	41	0.7	1.6	6.2	0.5	7.5	680	83	12.2
30 - 50	43	1.3	3.1	4.4	0.5	12.1	710	115.7	16.3
50 - 70	44	1.7	3.8	5.6	0.6	11	630	56.1	8.9
70 - 85	42	1.1	2.7	5.2	0.4	7.1	740	58.5	7.9
85 - 100	40	1.8	4.5	7.1	0.7	9.3	690	80.7	11.7

Note: The minimum and maximum values used for the multi-run spatial variability analysis corresponded to values one standard deviation (S.D) from the mean (*σ*).

Appendix 1b. Dunk Island golf course measured hydraulic properties – statistical data

Soil Layer *[cm]*	*σ* *Porosity* θ_s *[%]*	*S.D* *Porosity* θ_s *[%]*	*CV* *Porosity* θ_s *[%]*	*σ* *Wilting Point* θ_{wilt} *[%]*	*S.D* *Wilting Point* θ_{wilt} *[%]*	*CV* *Wilting Point* θ_{wilt} *[%]*	*σ* *Saturated Conductivity* K_{sat} *[cm d⁻¹]*	*S.D* *Saturated Conductivity* K_{sat} *[cm d⁻¹]*	*CV* *Saturated Conductivity* K_{sat} *[%]*
0 – 5	42	5.9	14.1	8.1	3.1	38.0	34.0	22.8	67.1
5 - 10	43	2.8	6.5	9.2	2.2	24.2	22.0	8.5	39.2
10 - 20	41	1.9	4.6	11.7	1.0	8.9	11.0	2.4	22.4
20 - 30	38	4.2	11.1	16.2	5.1	31.3	1.7	1.6	94.6
30 - 50	36	3.6	9.9	20.8	4.7	22.7	3.5	1.9	55.3
50 - 70	37	1.8	4.8	19.9	2.3	11.8	2.9	1.2	42.1
70 - 85	38	2.0	5.2	21.5	3.0	14.1	4.8	2.3	48.7
85 - 100	39	5.0	13.0	18.6	5.1	27.2	2.8	2.0	71.0

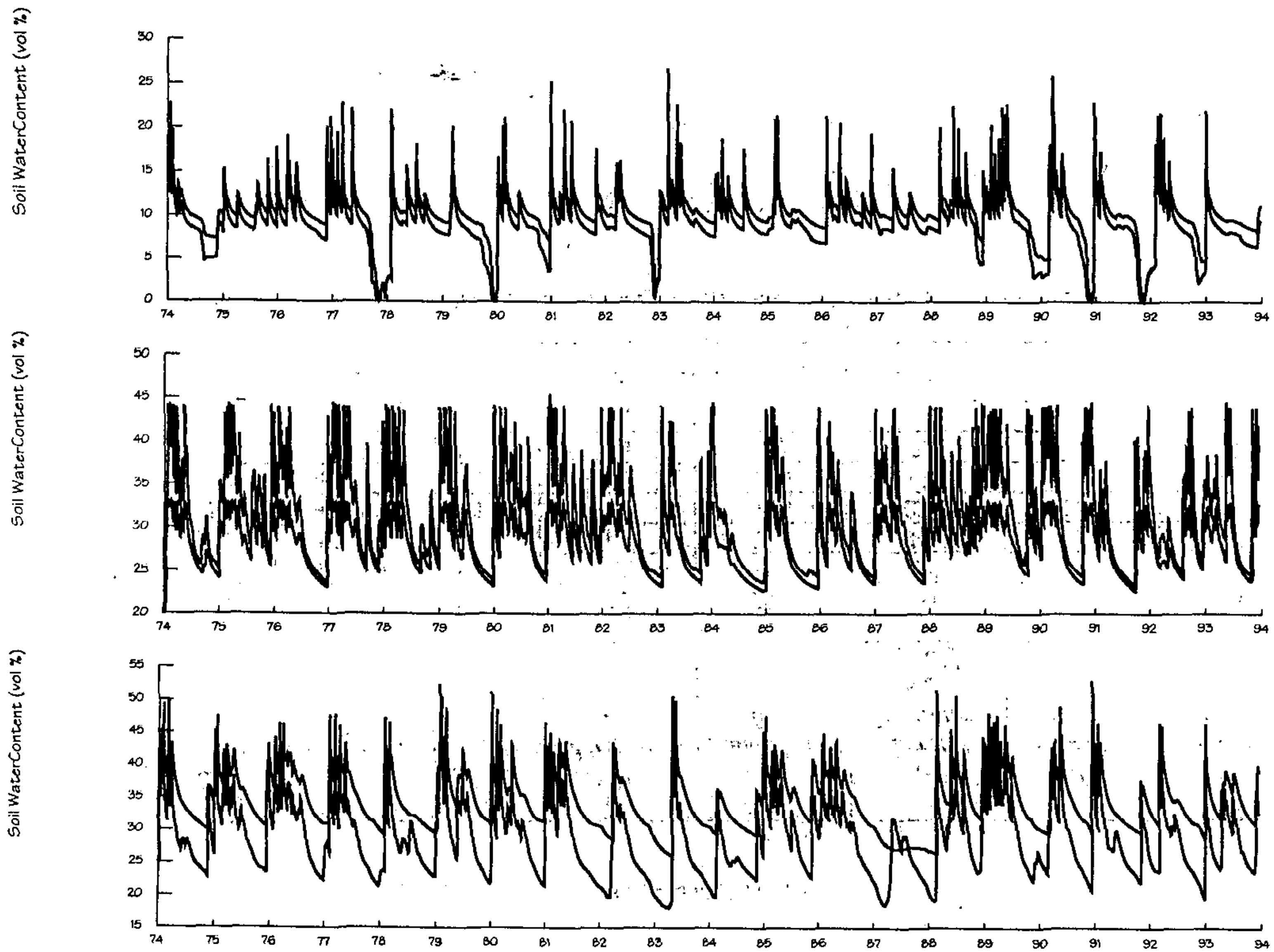
Note: The minimum and maximum values used for the multi-run spatial variability analysis corresponded to values one standard deviation (S.D) from the mean (*σ*).

Appendix 1c. Brampton Island golf course measured hydraulic properties – statistical data

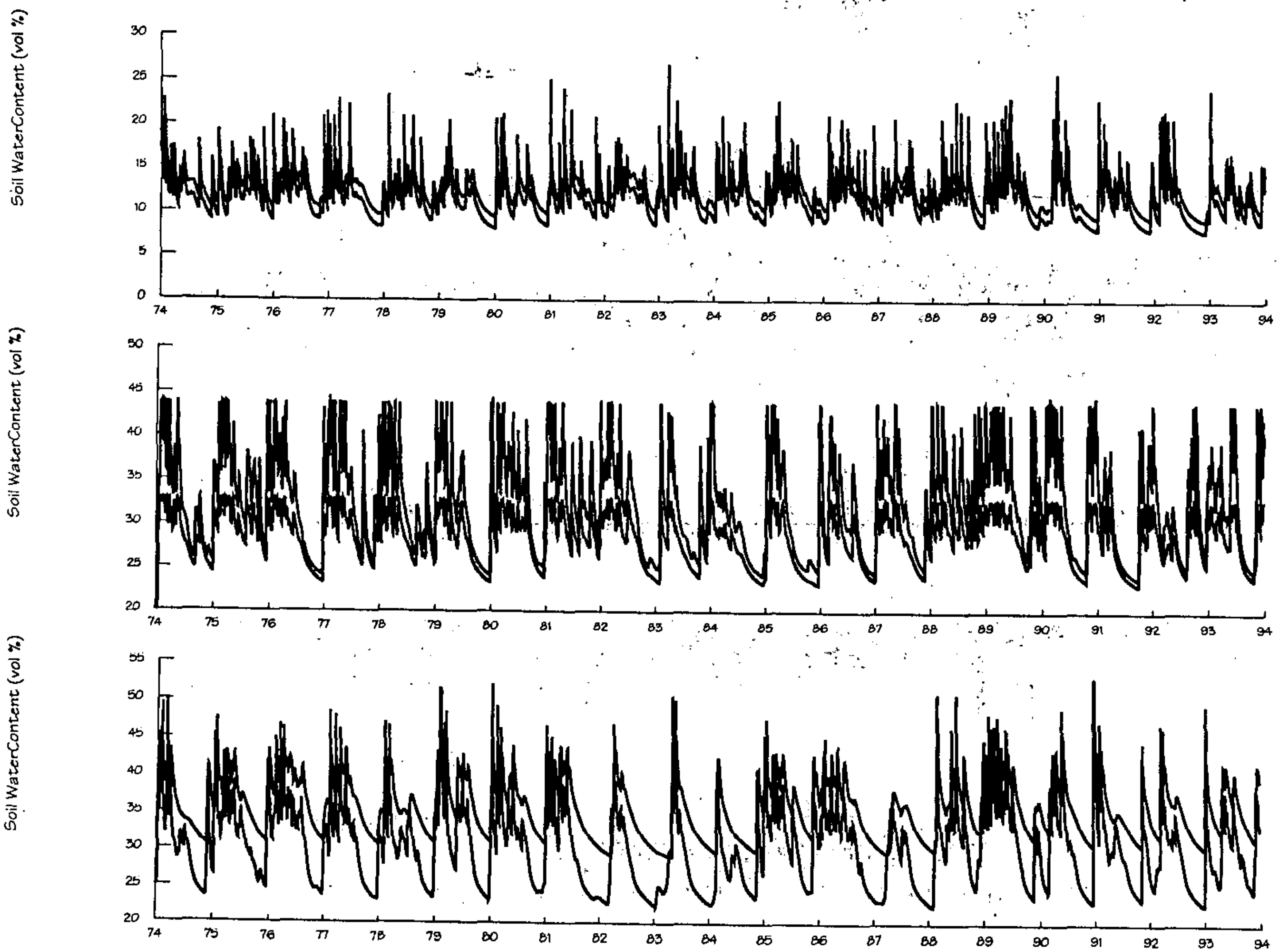
Soil Layer [cm]	σ Porosity θ_s [%]	S.D Porosity θ_s [%]	CV Porosity θ_s [%]	σ Wilting Point θ_{wilt} [%]	S.D Wilting Point θ_{wilt} [%]	CV Wilting Point θ_{wilt} [%]	σ Saturated Conductivity K_{sat} [cm d^{-1}]	S.D Saturated Conductivity K_{sat} [cm d^{-1}]	CV Saturated Conductivity K_{sat} [%]
0 – 5	42	2.8	6.7	8.4	3.3	39.7	51	22.3	43.8
5 - 10	39	2.8	7.1	9	3.2	35.2	35	13.2	37.7
10 - 20	42	1.5	3.5	8.3	1.1	13.3	44.6	40.0	89.7
20 - 30	45	1.8	3.9	11.5	3.7	31.8	22.2	17.6	79.4
30 - 50	49	2.1	4.2	12.8	3.5	27.6	15.4	14.2	92.3
50 - 70	46	3.0	6.5	20	4.6	23.2	2.2	1.4	64.6
70 - 85	48	2.5	5.3	17	3.2	19	8.9	7.4	83
85 - 100	50	4.1	8.2	15.5	3.5	22.9	12.7	12.1	95.4

Note: The minimum and maximum values used for the multi-run spatial variability analysis corresponded to values one standard deviation (S.D) from the mean (σ).

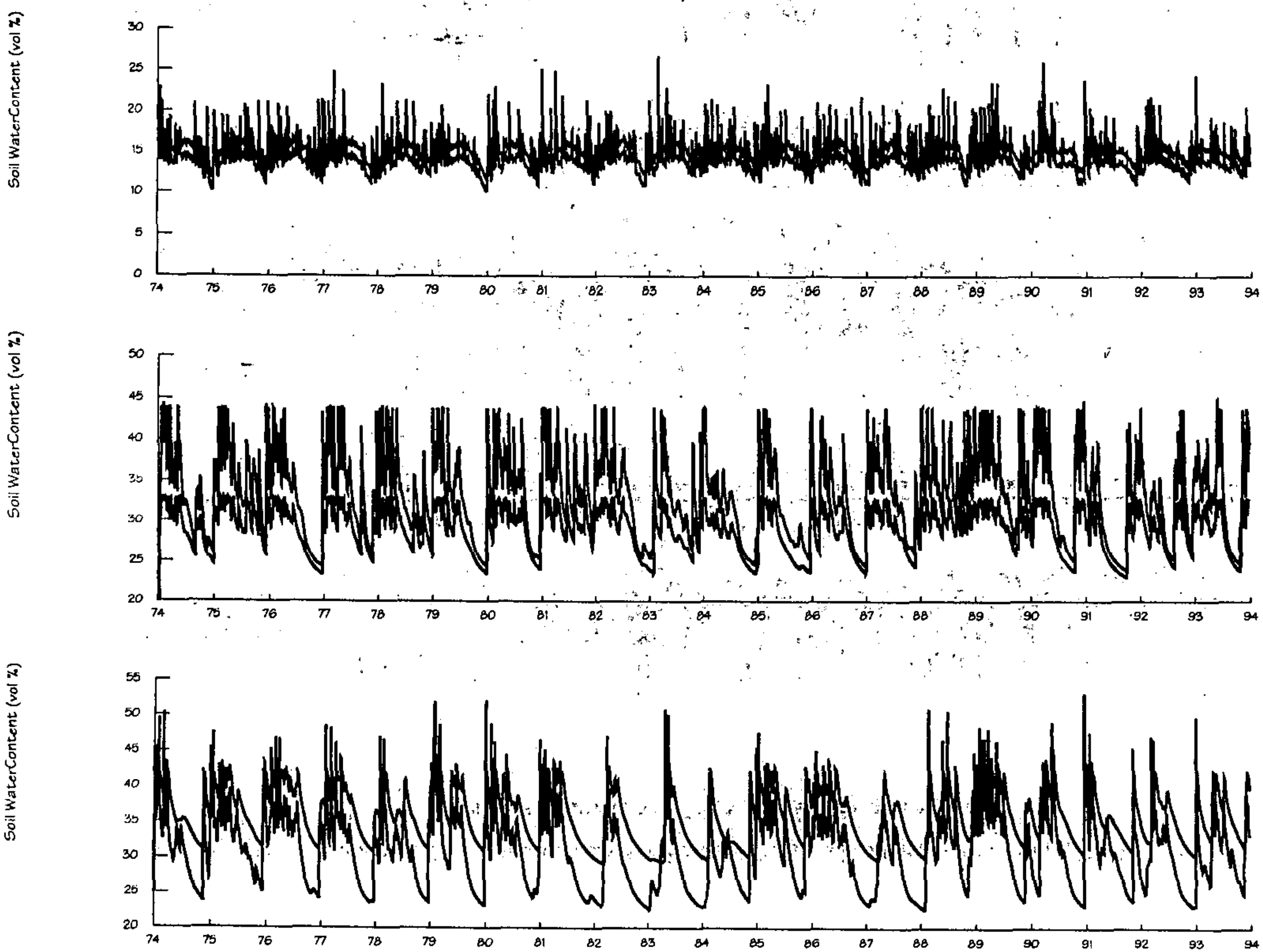
Appendix 2a. Simulated envelope of variation in soil moisture content for the 0.85 - 1.0 m cell at Great Keppel (top), Dunk (centre) and Brampton (bottom) Islands for irrigation = 0 % of total daily effluent.



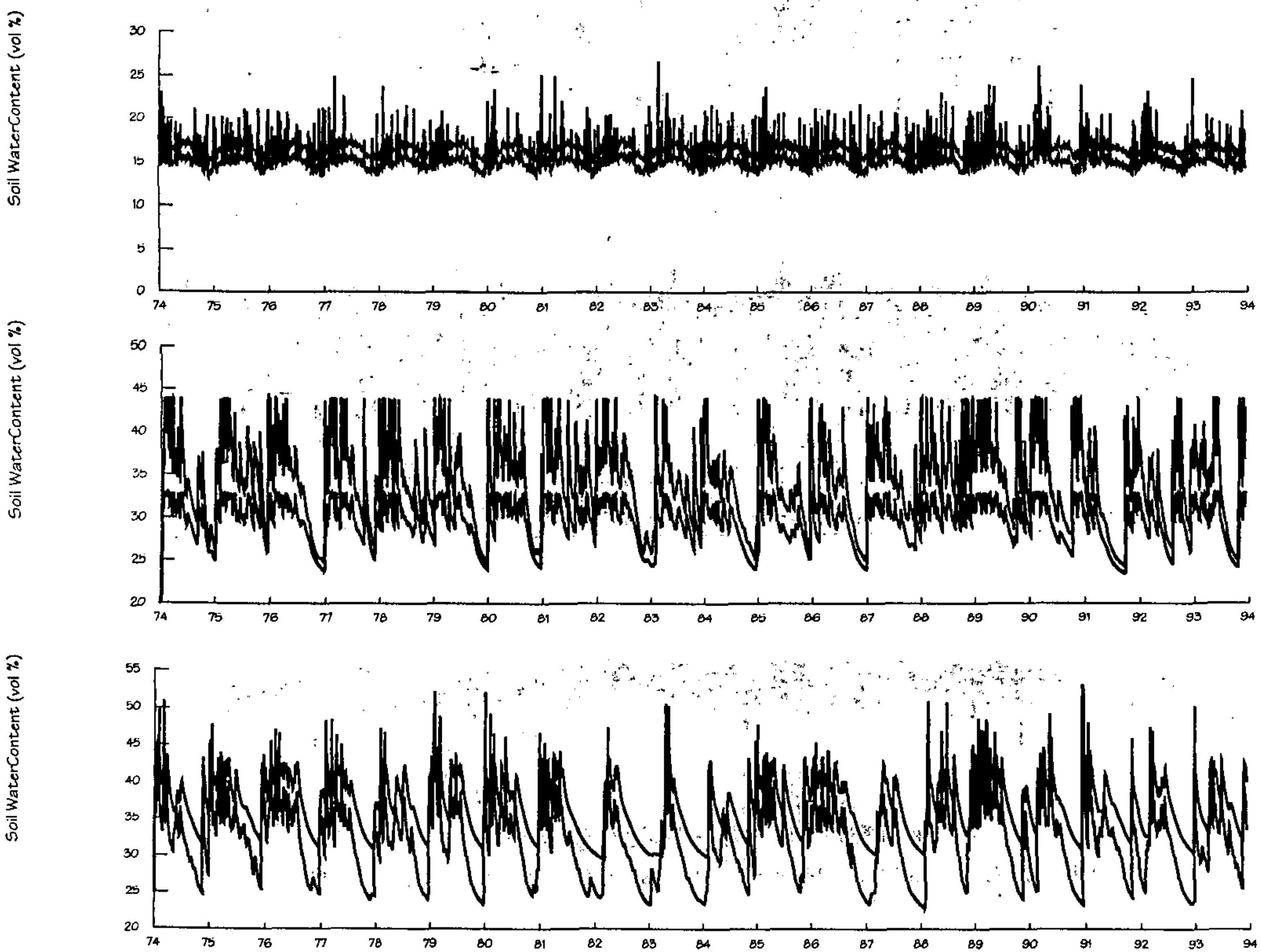
Appendix 2b. Simulated envelope of variation in soil moisture content for the 0.85 - 1.0 m cell at Great Keppel (top), Dunk (centre) and Brampton (bottom) Islands for irrigation = 25 % of total daily effluent.



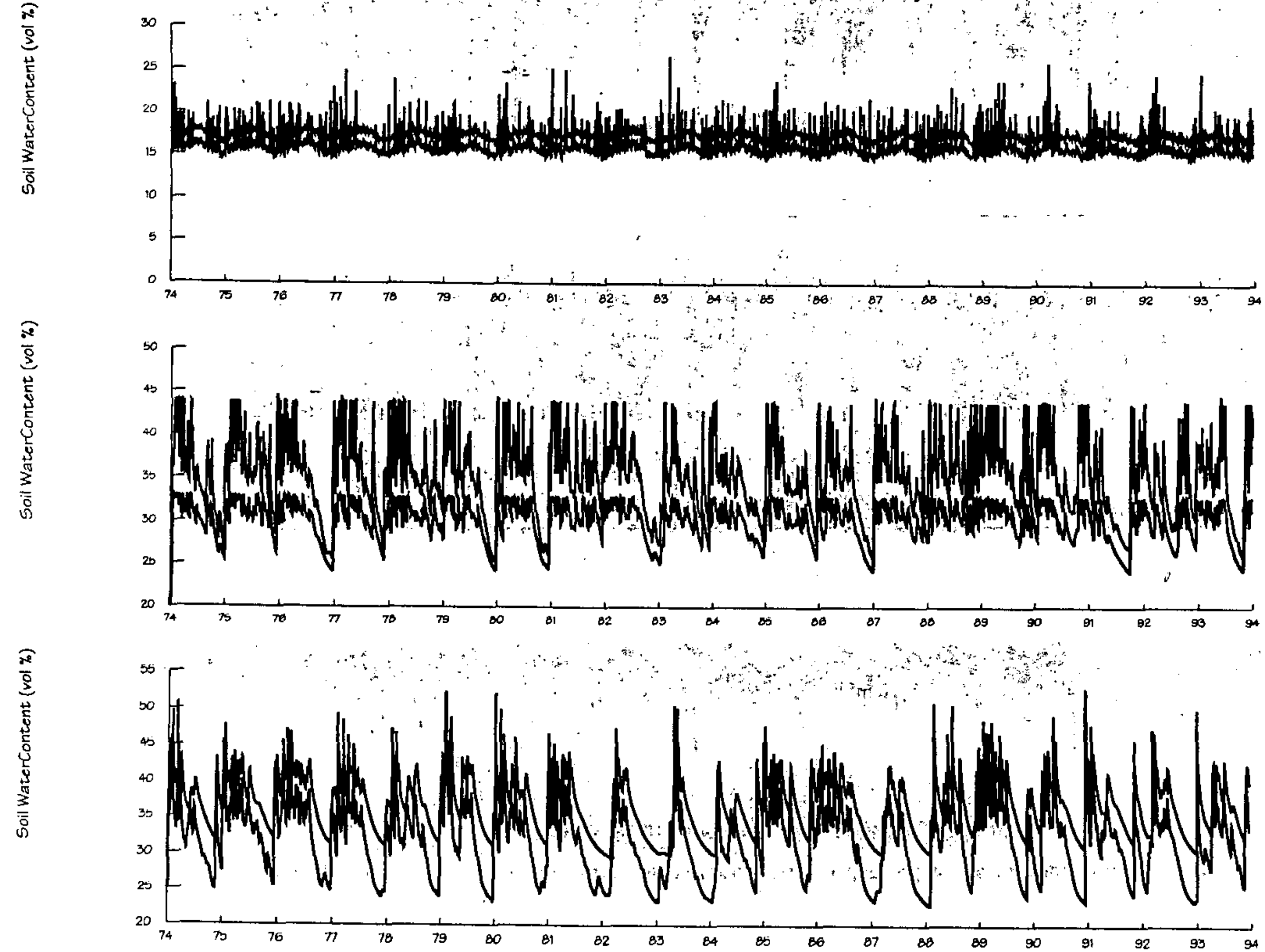
Appendix 2c. Simulated variation in soil moisture content for the 0.85 - 1.0 m cell at Great Keppel (top), Dunk (centre) and Brampton (bottom) Islands for irrigation = 50% of the total daily effluent.



Appendix 2d. Simulated envelope of variation in soil moisture content for the 0.85 - 1.0 m cell at Great Keppel (top), Dunk (centre) and Brampton (bottom) Islands for irrigation = 75 % of total daily effluent.



Appendix 2e. Simulated variation in soil moisture content for the 0.85 - 1.0 m cell at Great Keppel (top), Dunk (centre) and Brampton (bottom) Islands for irrigation = 100 % of total daily effluent.



Appendix 3. Parameter values used in the SOILN model.

General Process & Parameter	Parameter Description	Value	Unit	Source
1. Precipitation				
DEPWC	Mineral N concentration of rainfall	1.6 E-3	mg N L^{-1}	1
DEPFNH4W	Fraction of NH_4^+ in rainfall	0.95	-	1
DEPDRYA	Dry deposition of mineral N on canopy per unit of leaf area and which is taken up by leaves	0	g N m^{-2} d^{-1}	*A
2. Turfgrass Root Uptake				
ROOTDMIN	Lowest level of roots	1.1	m	6
ROOTDINC	Parameter determining *root depth as function of root biomass*	-0.06, -0.04	-	6*B
UPMA	Fraction of mineral N available for uptake at each time step	0.08	-	12
UPMOV	Compensatory N uptake from soil layers	1.0	-	10
	Distribution of the root density from soil surface to the root depth	Exponential	-	*A
RFRACLOW	Fraction of exponential function remaining below root depth	0.05	-	*A
3. Turfgrass Biomass Allocation				
WLAI	Specific leaf area	0.018	m^2 g DM^{-1}	4
AROOTN	Fraction of total growth allocated to roots	0.1	d^{-1}	7
ALEAF(1)	Coefficient for leaf area development as function of shoot biomass	0.06	-	25
ALEAF(2)	Coefficient for leaf area development as function of shoot biomass	-0.008	-	25
4. Turfgrass Leaf Assimilation				
EXTCOEF	Radiation extinction coefficient for the canopy	0.6	-	17
NLEAFN	Leaf N concentration in leaf at which minimum photosynthesis occurs	0.005	g N g DM^{-1}	11
NLEAFXG	Leaf N concentration in leaf at which maximum photosynthesis occurs	0.04	g N g DM^{-1}	7
PHOEFF	Radiation use efficiency at optimum temperature, water and N conditions	3.5	g DM MJ^{-1}	7
PTRANSM	Leaf transmission coefficient	0.1	-	18
PGRESP	Respiration growth conversion efficiency	0.72	-	19
PPMAX20(1)	Maximum leaf photosynthesis rate at optimal temperature, water and N conditions	54	kg CO_2 ha^{-1} h^{-1}	7
PPMAX20(2)	Rate of decline of maximum leaf photosynthesis with increased leaf area index	0.35	-	28

Appendix 3. (continued)

PHOTEMP(1)	Minimum daily mean air temperature for growth	0	°C	*A
PHOTEMP(2)	Daily mean air temperature for optimum growth	25	°C	*A
5. Turfgrass Respiration & Litter				
ALITTERS	Fraction of stem biomass lost as litter	0.026	d^{-1}	26
ALITTERL	Fraction of leaf biomass lost to litter	0.023	d^{-1}	26
ALITTERR(1)	Fraction of root growth lost as litter	0.03	d^{-1}	27
ALITTERR(2)	Fraction of root biomass lost as litter	0.03	d^{-1}	27
TEMBASP	Base temperature at which temperature effect = 1 for grass respiration	25	°C	*A
TEMQ10P	Response to a 10 °C soil temperature change for grass respiration.	2.6	-	20
6. Denitrification				
	Distribution of the denitrification rate below soil surface	Exponential	-	14
DENDEPTH	The depth where the denitrification capacity ceases	0.5	m	5, 16
DFRACLOW	Fraction of the exponential function remaining below the depth where the denitrification activity ceases	-0.05	-	*A
DENPOT	Potential rate of denitrification:			
	Clay-based soil with high residues (Brampton Island, Dunk Island)	0.15	$g\ N\ m^{-2}\ d^{-1}$	5
	Sandy-based soil with high residues (Great Keppel Island)	0.1	$g\ N\ m^{-2}\ d^{-1}$	21
DENHS	Half saturation constant in function for NO_3^- concentration effect	10	$mg\ N\ L^{-1}$	22
TEMQ10D	Response to a 10 °C soil temperature change for denitrification – Brampton and Dunk Island	10		13
	Response to a 10 °C soil temperature change for denitrification – Great Keppel Island	1.6		13
7. Mineralisation & Immobilisation				
CNORG	C-N ratio micro-organisms and humified products	6	-	2, 3
CPLANT	C content of biomass when lost as litter	0.45	$g\ C\ g\ DM^{-1}$	8
ABOVEK	Fraction of N and C in surface residues that are converted to litter	0.1	d^{-1}	3
ABOVELN	Fraction of N in above ground residues that are leached every day	0	d^{-1}	*A
ABOVELC	Fraction of C in above ground residues that are leached every day	0	d^{-1}	*A
LITK	Litter specific decomposition rate under grassland	8E-3	d^{-1}	3
HUMK	Humus specific decomposition rate under grassland	1.5E-5	d^{-1}	29
NITK	Specific nitrification rate	0.09	d^{-1}	9
NITR	NO_3^- - NH_4^+ ratio in nitrification function	1	-	10

Appendix 3. (continued)

LITHF	Litter carbon humification fraction	0.35	-	23
LITEFF	Efficiency of the internal synthesis of microbial biomass and metabolites in litter	0.4	-	3
8. Soil Abiotic Response				
MOSM	Coefficient in soil-water function for mineralisation-immobilisation and nitrification processes	1	-	*A
MOSSA	Saturation activity in soil moisture response function for mineralisation-immobilisation and nitrification processes	0.6	-	12
MOS(1)	Difference in moisture contents between wilting point and the minimum moisture content at which the moisture content response function has a value of unity for mineralisation-immobilisation and nitrification processes	10	Vol %	12
MOS(2)	Difference in moisture contents between saturation and the maximum moisture content at which the moisture content response function has a value of unity for mineralisation-immobilisation and nitrification processes	8	Vol %	12
TEMBAS	Base temperature at which temperature effect =1 for mineralisation-immobilisation, denitrification and nitrification processes	25	°C	*A
TEMQ10	Response to a 10 °C soil temperature change for the mineralisation-immobilisation and nitrification processes.	1.6	-	24
DEND	Coefficient in function for soil-water content / aeration effect on denitrification	2	-	13*B
MOSDEN	Water content range in function for soil-water content / aeration effect on denitrification	15	Vol %	15*B

Sources: 1. Furnas et al. (1994); 2. Robertson et al. (1993); 3. Probert et al. (1998); 4. Boot and den Dubbelden (1990); 5. Pu et al. (1999); 6. Devitt (1989); 7. Hodgkinson et al. (1989); 8. Ghannoum and Conroy (1998); 9. Strong and Cooper (1992); 10. Blomback and Eckersten (1997); 11. van Keulen et al. (1989); 12. Johnsson et al. (1987); 13. Smith et al. (1998); 14. Luo et al. (1998); 15. Ruz-Jerez et al. (1994); 16. Burford and Bramner (1975); 17. Madakadze et al. (1998); 18. Faurie et al. (1996); 19. Bunce (1995); 20. Jellinghaus et al. (1996); 21. Davidsson and Leonardson (1997); 22. Johnsson et al. (1991); 23. Bloemhof and Berendse (1995); 24. MacDuff and White (1985); 25. Wu et al. (1998); 26. Sheehy et al. (1980); 27. Thornley and Verberne (1989); 28. Topp and Doyle (1996); 29. Jansson and Andersson (1988).

Notes: *A Assumed value; *B Adapted from source

Appendix 4. Initial conditions of nitrogen, carbon and biomass pools of turfgrass and soil for the SOILN model.

Parameter	Parameter Description	Value	Unit	Source
NO_3 (1-8)	Nitrate-N in defined soil layers 1-8	0.13, 0.13, 0.39, 0.32, 0.3, 0.3, 0.27, 0.29	g N m^{-2}	1
NH_4 (1-8)	Ammonium-N in defined soil layers 1-8	0.1, 0.1, 0.15, 0.15, 0.24, 0.24, 0.18, 0.11	g N m^{-2}	1
NH (1-8)	Humus-N in defined soil layers 1-8 under turfgrass	105, 120, 195, 180, 270, 180, 115, 115	g N m^{-2}	2
CL(1-4)	Litter carbon in defined soil layers 1-4 under turfgrass	21, 17, 14.4, 14.4	g C m^{-2}	3
NLIT(1-4)	Total-N in litter in defined soil layers 1-4 under turfgrass	5.2, 4, 2.3, 2.3	g N m^{-2}	3
LEAFN	Initial value of N in turfgrass leaves	3.8	g N m^{-2}	4
STEMN	Initial value of N in turfgrass stems	1.3	g N m^{-2}	4
ROOTN	Initial value of N in turfgrass roots	0.7	g N m^{-2}	4
LEAFW	Initial value of turfgrass leaf biomass	135	g DM m^{-2}	5
STEMW	Initial value of turfgrass stem biomass	45	g DM m^{-2}	5
ROOTW	Initial value of turfgrass root biomass	25	g DM m^{-2}	5

Sources: 1. Adapted from Probert et al. (1998) using mean measured bulk density at Brampton Island; 2. Measured at Brampton Island; 3. Adapted from Robertson et al. (1993) using mean measured bulk density at Brampton Island; 4. Wilman et al. (1994); 5. Topp and Doyle (1996).