Nitrogen balance for an agroforestry system irrigated with a saline, high nitrogen effluent

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Abstract
Land disposal is commonly used for urban and industrial wastewater, largely due to the high costs involved in alternative treatments or disposal systems. However, the viability of such systems depends on many factors, including the composition of the effluent water, soil type, the plant species grown, growth rate, and planting density. The objective of this study is to establish whether land disposal of nitrogen (N) rich effluent using an agroforestry system is sustainable, and determine the effect of irrigation rate and tree planting density on the N cycle and subsequent N removal.

We examined systems for the sustainable disposal of a high strength industrial effluent. The challenge was to leach the salt, by using a sufficiently high rate of irrigation, while simultaneously ensuring that N did not leach from the soil profile. We describe the N balance for two plant systems irrigated with effluent, one comprising *Eucalyptus tereticornis* and *Eucalyptus moluccana* and a Rhodes grass (*Chloris gayana*) pasture, and the other, Rhodes grass pasture alone.

Nitrogen balance was assessed from N inputs in effluent and rainfall, accumulation of N in the plant biomass, changes in soil N storage, N loss in run-off water, denitrification and N loss to the groundwater by deep-drainage.

Biomass production was estimated from allometric relationships derived from yearly destructive harvesting of selected trees. The N content of that biomass was then calculated from measured N content of the various plant parts, and their mass. Approximately 300 kg N/ha/yr was assimilated into tree biomass at a planting density of 2500 tree/ha of *E. moluccana*. In addition to tree assimilation, pasture growth between the tree rows, which was regularly harvested, contributed substantially to N uptake. If the trees were harvested after two years of growth and grass harvested regularly, biomass removal of N by the mixed system would be about 700 kg N/ha/yr.

The results of this study show that the current system of effluent disposal is not sustainable as the nitrate leaching from the soil profile far exceeds standards set out by the ANZECC guidelines. Hence additional means of N removal will need to be implemented. Biological N removal is an area that warrants further studies as it is aimed at reducing N levels in the effluent before irrigation. This will complement the current agroforestry system.

Key Words
Nitrate, ammonium, leaching, volatilisation, denitrification, agroforestry.

Introduction
The disposal of industrial and urban wastewater has become an issue of public concern during recent years as increased environmental awareness has fundamentally changed Australian community attitudes. For years being viewed as a waste product, effluent is now being seen as a potential resource with a wide range of useful land based applications, including industrial and domestic reuse, and the irrigation of recreational and agricultural land and tree plantations (Hopmans *et al*. 1990; Myers *et al*. 1999; Guo and Sims 2000; 2003). The objective of this study is to determine whether land disposal of nitrogen (N) rich effluent using an agroforestry system is sustainable.

An important consideration regarding effluent-irrigated plantations is the impact that this practice may have on soil quality and site productivity in the short- and long-term. The potential for accumulation of nitrates, salt, and other toxins, requires close monitoring to minimise the effects on soil structure, permeability and porosity. Leaching of nitrate or salt into the groundwater may reduce drinking water

quality and environmental values, whilst runoff of phosphates and nitrates into rivers can promote undesirable toxic algal blooms (Stewart *et al*. 1990).

This study evaluates deep leaching high strength industrial effluent with the objective of using the soil as a repository for applied N, while allowing the vertical movement of salts into the groundwater and ultimately to a nearby river. A principal challenge to be addressed in this effluent disposal system is to leach the applied salt using a sufficiently large irrigation rate, while ensuring N uptake by the plants is sufficient to prevent leaching from the soil profile. To establish an acceptable off-site outcome, the N concentration in both the leachate reaching the groundwater and runoff reaching the adjacent river must be below the ANZECC guideline for freshwaters of 10 mg/L (ANZECC 1992). To achieve this set level, both plant N uptake and N loss through denitrification must be maximised.

Methods

Overview of experiment

The study was conducted on a 171 ha agricultural property 8 km to the south west of Beaudesert in southeast Queensland, Australia (latitude: 28°1'S, longitude 152°55'E). The field trial site was located on the alluvial floodplain of the Logan River. Soil at the site can be described as a brown mesotrophic haplic dermosol (Isbell 1996). The soil is characterised by a sandy loam A horizon to 60cm and a less permeable sandy clay loam B horizon.

The trees employed in this study were planted in March 2001 and were comprised of two species of eucalypt, *Eucalyptus moluccana* and *Eucalyptus tereticornis*. These species were selected for their high levels of salt tolerance and widespread use in agroforestry and saline land reclamation. A total of 1344 trees were planted into 32 plots at one of two planting densities; 1250 trees/ha (low) or 2500 trees/ha (high). Two irrigation rates were applied over the trial, with half of the trees receiving 25 mm per week or 1300 mm per year (low irrigation) and the other half-receiving 50 mm per week or 2600 mm per year (high irrigation). Irrigation was initially via an overhead irrigation system, later changed in August 2003 to under-tree sprinkler irrigation.

The effluent utilized is characterised by high salinity, with electrical conductivity (EC) readings in excess of 6 dS/m. Nitrogen input was also high, mainly in the forms of ammonium and nitrate. The effluent had a neutral pH.

Nitrogen balance

The N balance can be described in terms of inputs, outputs and changes in storage that occur in various N pools as follows:

$$N_{irr} + N_{rain} = N_{plant} + N_I + N_{ro} + N_v + N_{den} + N_s \qquad \text{Eq.1}$$

Where:
N_{irr} = effluent input
N_{rain} = rainfall input
N_{plant}= tree and grass uptake
N_I = inorganic N leached
N_{ro} = N lost in runoff (total dissolved N + total particulate N + mineralisable N in bedload)
N_v = N volatilised
N_{den} = N denitrified
N_s = change in soil N storage

Effluent input

To establish the composition of the effluent being irrigated, a 50 mL sample of irrigation water was collected weekly during irrigation events. On collection, samples were titrated with 1m HCl to a pH of 3, and then stored in airtight containers in ice, before being filtered to 0.45 µMol and analysed for total Kjeldahl N (TKN), ammonium and nitrate. The organic nitrogen captured on the filter paper was measured using a LECO CNS 2000 combustion analyser at 1100°C calibrated with EDTA and added into the TKN calculations. It should be noted that for laboratory analysis purposes TKN is a test performed

that is made up of both organic N and ammonia. For this reason nitrate concentrations need to be added to TKN when establishing total N concentrations of the effluent.
A major limitation to this study was a lack of constant irrigation of effluent on to the trial site. Due to technical difficulties with the irrigation system and drought conditions, which resulted in water restrictions, the irrigation was sporadic and unpredictable hence a true measure of yearly irrigation rates were impossible to determine. However with the use of some periods of prolonged irrigation and simultaneous monitoring during both summer and winter, critical aspects of the N balance were determined.

Consequently, a model (Figure 1) was developed to compensate for the lack of a constant irrigation rate. This model takes into account yearly rainfall, potential evapo-transpiration (ET) and the desired irrigation rate required to leach the salt applied in the effluent (EC 6 dS/m). For the purpose of this model, an EC of 12 dS/m was set for drainage water, thus ensuring salt did not build up in the soil profile. Given that yearly rainfall and ET is known, an optimal irrigation rate was determined (Figure 1) as the difference between rainfall and ET, multiplied by 2, to ensure such a leaching fraction occurred.

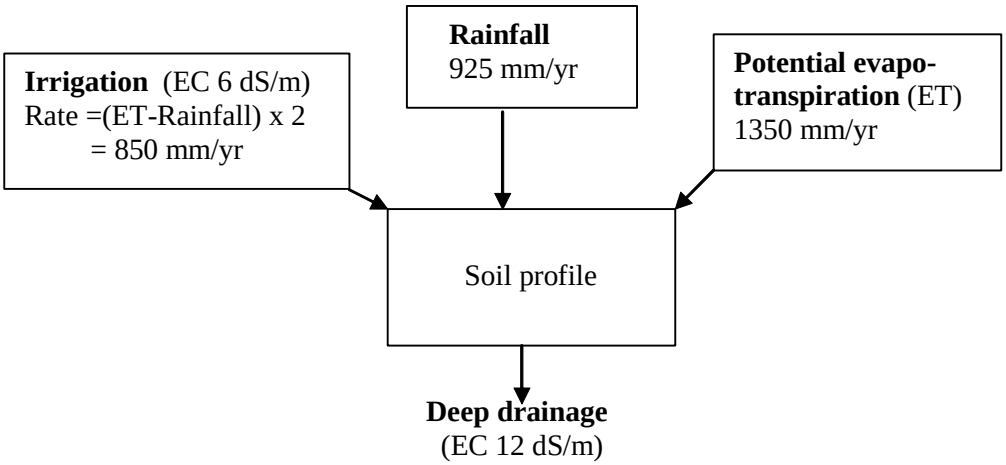

Figure 1. Proposed model of the salt and water balance for the study site to ensure an adequate leaching fraction occurred.

Tree and grass N uptake
Uptake was assessed in a three-year-old plantation of the two difference eucalypt saplings species, (*E. tereticornis* and *E. moluccana*) planted at the two densities (1250 and 2500 trees/ha). The trial was established as a factorial (2 species x 2 planting densities) with 4 replications of each treatment.

Biomass production for eucalypts was determined by destructive harvest of selected trees on a yearly basis. The N content of the various plant parts when then calculated using a LECO CNS 2000 combustion analyzer at 1100°C calibrated with EDTA, and their mass used to determine total tree nitrogen content. The attributes of tree height and girth were used to generate allometric relationships. These relationships enabled calculation of biomass and nutrient assimilation capabilities of the two tree species, and the effects of planting density on tree growth.

Pasture growth between the tree rows, and in the separate pasture plot was assessed using the Botanal technique (Tothill *et al*. 1992; Hargreaves and Kerr 1992; McDonald *et al.* 1996). Herbage sub-samples (*ca. 200-400*g) where collected and dried at 55°C in a dehydrator for three days before being reweighed and a regression curve generated. This regression curve was then used to determine inter row pasture yields for the eucalpt trial. The N content was also determined for the two main grass species Rhodes grass (*Chloris gayana*) and Couch grass (*Cynodon dactylon)*. This enabled comparison between the N assimilation by pasture along and combined trees and pasture system.

The data was analysed by a one-way ANOVA, when a significant difference was found pos-hoc tests were carried out.

Volatilisation
Loss of N by volatilisation of ammonia during the spray application of effluent was determined using rainfall collection jars distributed at varying distances around a sprinkler head. Sulphuric acid was added to each jar to reduce the pH of the collected effluent to 2-3 to prevent further volatilization. A water sample was bled off the irrigator at commencement of the experiment and every 10 minutes thereafter. After an hour all jars were collected. The water was filtered to 0.45 µMol and stored on ice before being analysed in the laboratory for ammonium using distillation and titration. Chloride levels were also determined using the mercuric thiocyanate method (Anon 1995), as a means of determining the relative level of evaporation and subsequent concentration effect.

Inorganic N leached, denitrification & soil N storage
Inorganic N leaching, denitrification and the change in soil N were calculated as the difference between inputs, biomass accumulation and outputs. To determine the amount of nitrate being leached from the system, ceramic cup soil solution samplers were used. Solution samplers were installed 1.5m below the active root zone for each treatment, thus giving an indication of the concentration of nitrate in the drainage water passing from the soil to the shallow aquifer. Potential denitrification losses is currently being evaluated in a glass house trial using soil columns. Redox probes and solution samplers are being utilised to monitor any nitrate losses as a result of denitrification.

Results and Discussion

The effluent used for irrigation contained large concentrations of ammonium and nitrate (Table 1). With an irrigation rate of 850 mm/yr (Figure 1), the total N input from the effluent would be 4,675 kg N/ha/yr (Calculation 1). This highlights the need to maximize N removal in order to minimise the amount of nitrate being leached to the ground water.

Table 1. Average effluent composition (n=20).

Electrical conductivity (dS/m)	Total Kjeldahl N (mg/L)	Nitrate (mg/L)	Ammonium (mg/L)
6	450	100	300

A small amount of additional N is added to the system within rainwater. Studies by Douglas (1968) and Ayers and Manton (1991) found nitrate and ammonium concentrations to be 6 µMol/L within the rainwater from near coastal locations within Australia. This has to be taken into consideration when calculating the total N inputs for the system. The average rainfall for this area is 925 mm/yr therefore adding a further 1.55 kg N/ha/yr (Calculation 2).

The percentage of ammonia within the effluent is strongly dependent on pH. The effluent in this study had a pH of 7.0 and was expected to have 8 to 10% of the total ammoniacal nitrogen (TAN = NH_4^+-N + NH_3-N) in the ammonia form (Denmead *et al.* 1982; Jayaweera and Mikkelsen 1990; Ruxton 1995). Therefore, only a small proportion of the TAN could potentially be lost to the atmosphere during spray irrigation. This study found ammonia volatilisation losses of 2 to 5% equivalent to 156 to 390 kg N/ha/yr at the proposed irrigation level (Calculation 3). The variation in ammonia loss during irrigation is attributed to air temperature, relative humidity, drop diameter, irrigation pressure, spray velocity, effluent TAN concentration, and pH (Denmead *et al.* 1982; Brunke *et al.* 1988; Sharpe and Harper 1997).

Ammonia volatilisation within the soil is also strongly affected by pH. A high pH increases the concentration of ammonia present in the soil solution and soil air, increasing the potential for ammonia loss from soil. Jewitt (1942) demonstrated this effect, finding ammonia losses of 87%, 13% and 0% when ammonium sulphate was applied to soils of pH 10.5, 8.6 and 7.0 respectively. As the surface soil of this study site has a pH in the range 3.5 to 4, no loss of N by ammonia volatilisation was anticipated once the effluent reaches the soil.

In this study we examined N removal by pasture alone in comparison to a system employing both trees and pasture. There was no significant difference in pasture production per unit of pasture area, regardless of the presence of trees (15 t/ha/yr dry matter in pasture plot; 16.5 t/ha/yr dry matter between tree rows), though the proportion of the ground area covered by grass decreased as the trees reached canopy closure. This result indicates that a system utilising both trees and pasture would provide maximum uptake of N.

A linear relationship between N assimilation and planting density was found (Figure 2). It is considered that as the trees reach maturity, competition for nutrients will play a more significant role in tree biomass production. In comparing the two eucalypt species (Figure 2), *E. moluccana* was found to have a significantly greater (P<0.05) total N content, due to a larger biomass production, than *E. tereticornis*.

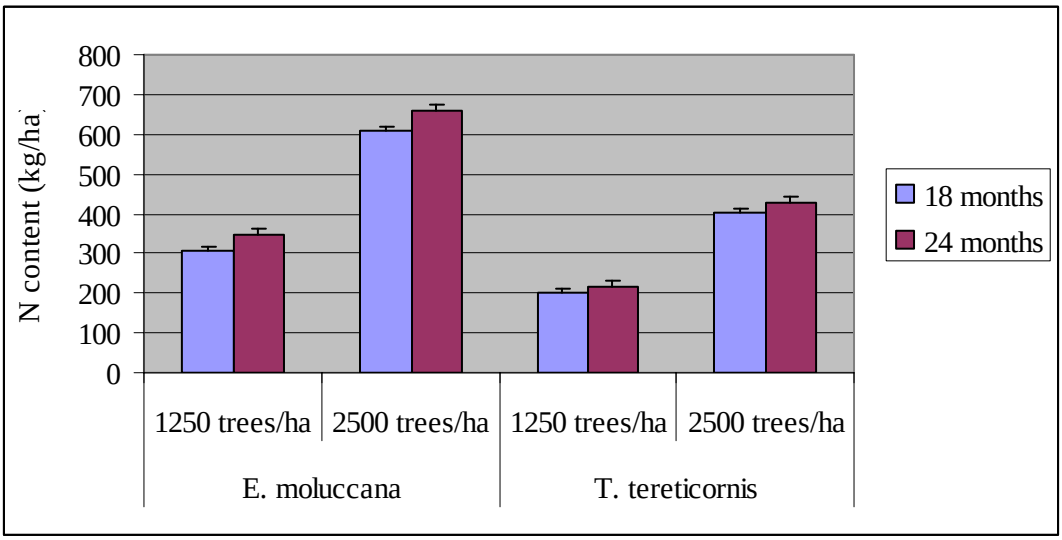

Figure 2. Effect of tree species and planting density on N uptake by *Eucalyptus* spp. irrigated with high strength industrial effluent, 18 and 24 months after planting. (*E. moluccana 1250* trees/ha vs low 2500 trees/ha p<0.01; *E. tereticornis* 1250 trees/ha vs low 2500 trees/ha p<0.05; *E. Moluccana* vs *E. tereticornis* p<0.01)

Total tree N per hectare per annum was derived from the data obtained at 24 months of growth. For *E. moluccana*, over 300 kg N/ha/yr was assimilated into tree biomass at the higher planting density.

In addition to tree assimilation, pasture growth between the tree rows, which was harvested every three months, contributed greatly to N uptake. In 2003 inter row pasture assimilated 390 kg N/ha/yr. If the trees were harvested at two years of growth and grass harvested regularly, biomass removal of N by the mixed system was estimated to be approximately 700 kg N/ha/yr.

No runoff was observed during the trial and so no N was lost via surface runoff or soil erosion.

Using the N balance approach detailed in equation 1, the sum of all the inputs was 4,676 kg N/ha/yr. While the sum of the outputs was 1,090 (700 +390) kg N/ha/yr. This left a discrepancy of 3,586 kg N/ha/yr suggesting surplus N would either be denitrified, stored in the soil and/or leached to groundwater.

Soil storage of N in humus can only increase through cycling of organic matter, which is a slow process and is anticipated to only account for a small proportion of this excess nitrogen (Hart *et al*. 1994). Nitrogen stored in the soil as nitrate will be subject to leaching. Soil analysis will be conducted at the completion of this study to quantify the change in soil storage.

The potential for losses from the system via denitrification is currently being evaluated using soil column experiments. Preliminary data suggests N losses via dentrification are approximately 100 kg N/ha/yr. Onset of denitrification was slow, most probably due to the low biochemical oxygen demand (BOD) of the effluent. However, higher rates of N loss via denitrification could be achieved by manipulating the irrigation rate. This could be achieved by employing fewer but larger periods of irrigation to create waterlogging conditions, which would favour denitrification. Care needs to be taken that these periods of waterlogging don't have a detrimental effect on tree and pasture growth.

Movement of water downward through the soil profile causes the leaching of nitrate. The magnitude of N loss is proportional to the concentration of nitrate in soil solution and the volume of leaching water. Assuming the excess 3,586 kg N/ha/yr was accounted for by leaching of N to the ground water, the nitrate-N concentrations within the leachate would be approximately 530 mg/L. This far exceeds the 10 mg/L maximum nitrate level for potable water set by the ANZECC guidelines (Anon 1992). In order to

reduce the leaching of nitrate from the system to within these guidelines, denitrification will need to be maximised given that plant uptake has reached maximum levels. Even under the modest irrigation rates used on the trial to date, the nitrate concentration in soil solution collected at 1.5 m ranged from 26 to 290 mg N/L. The use of solution samplers involves some uncertainties due to the likelihood of channeling of water directly from soil surface to the vicinity of the cup, and also the distortion of water flow through the soil as a result of water potential applied (Talsma *et al*. 1979) or bypassing of water due to local heterogeneities (Goulding and Webster 1992).

The results of this study show that the current system of effluent disposal is not sustainable as the nitrate leaching from the soil profile far exceeds standards set out by the ANZECC guidelines. Hence additional means of N removal will need to be implemented. Biological N removal is an area that warrants further studies as it is aimed at reducing N levels in the effluent before irrigation. This will complement the current agroforestry system.

Conclusion

Nitrate losses from the system by leaching, far exceeded standards set by the ANZECC guideline for freshwater. In order to achieve these set N levels, losses by volatilisation and denitrification must be maximised as it is believed that plant N uptake has already reached maximum levels.

By raising the pH of the current effluent to pH>9 it is possible that larger quantities of N may be lost during irrigation. However, this raises the issue of ammonium and ammonia toxicity having a detrimental affect on plant growth. A further study into this option is currently underway.

A glasshouse trial is currently underway to investigate the potential quantity of N loss from the system by denitrification. Greater N losses from the system could be achieved by manipulating the irrigation rate to induce period of waterlogging, which favours denitrification. Alternative treatment options, such as biological nitrogen removal, may also be required to reduce N loading and improve the sustainability of the system.

Appendix 1. Calculations

Calculation 1. N added to system within irrigation water	Volume irrigation added = 0.850×10^4 m^3 or 0.85×10^7 L/ha This has a N concentration = 550 mg/L = $0.85 \times 10^7 \times 550$ mg N/ha = 467.5 x 10 kg N/ha = 4,675 kg N/ha/yr
Calculation 2. N added to system within Rainwater	6 μMol/L of NH_4^+ and NO_3 so rainwater contains = 12 μMol N/L or 12 $\times 10^{-6}$ Mol/L g N/L = (N concentration) x (atomic weight N) $12 \times 10^{-6} \times 14.0067 = 168 \times 10^{-6}$ g/L Rain upon field = 0.925×10^4 m^3 = 0.925×10^7 L N within Rainfall = $1.68 \times 10^{-6} \times 0.925 \times 10^7$ g/ha/yr = 1.55 x 10 g N/ha/yr = 1.55 kg N/ha/yr
Calculation 3. N losses by volatilisation	0.3 g NH_4^+ per L Therefore 7800 kg/ha/yr NH_4^+ is applied 2% loss by volatilisation = 0.02 x 7800 = 156 kg N/ha/yr 5% loss by volatilization = 0.05 x 7800 = 390 kg N/ha/yr

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