FATE AND TRANSPORT OF NITROGEN FROM WASTEWATER IRRIGATED TO LAND ADJACENT TO A SENSITIVE HARBOUR

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ABSTRACT (500 WORDS MAXIMUM)

As part of the reconsenting of the Omaha Wastewater Treatment Plant (WWTP) land discharge system, Watercare Services Ltd commissioned a study to understand the fate and transport of nutrients and the effects on the environment. The discharge of the treated wastewater is to two irrigation blocks, one on Watercare land (eucalypt and native bush) and the other being the Omaha Golf Course. While the discharge is to land, ultimately the water is transported through the groundwater to the Whangateau Harbour, which is a high quality waterway. Understanding the fate and transport of nutrients to maintain the harbour water quality was an important aspect for the stakeholders of this project.

We focussed on nitrogen transformations in soil beneath the irrigation sites. We measured a range of biochemical rates in soil cores to understand nitrate loss (denitrification) and nitrate production (nitrification). We also made additional chemical measurements to determine the inorganic nitrogen concentrations in pore water, the amount of readily mineralizable carbon, and redox potential (Eh) at the boundary with the groundwater table. We also commissioned other experts to estimate nitrogen uptake and immobilization in the eucalypt stand, and golf course, respectively.

A parallel study used a variety of geophysical measurements on the strata between the irrigation sites and the harbour to construct a model of groundwater flows and travel times. We used their soil-water leaching loss algorithms together with rate measurements derived from our biochemical testing and estimates of plant uptake/immobilization to construct a stochastic model of nitrogen loss from irrigated wastewater within the unsaturated soil profile. We also made estimates of likely N losses in saturated organic strata using an analytical solution that predicted N concentration as a function of conservative rate constants and residence time.

Our results showed that the peat soils on the Jones Rd sites had measurable in-situ denitrification throughout the soil profile. Denitrifying enzyme activity (DEA, indicator of size of denitrifying populations) was highest in the surface layers and declined with depth. On the golf course sites, high DEA and measurable in-situ denitrification activity occurred within the surface turf layer but declined to trace levels in the sand substrate beneath. Redox potential (Eh) at the boundary with the groundwater table was generally consistent with the published range of values associated with denitrification.

The modelling demonstrated that leached loads and concentrations were much higher for the golf course than the Jones Rd sites. This was due to a combination of high loads (in summer) and shorter residence times within a 15-30 cm thick ‘active' biological zone. Using conservative assumption for rate constants in the saturated zone (derived from measurements), the model predicted only trace concentrations (<0.1 g N/m^3) would enter the harbour. A sensitivity analysis of residence times showed that 200 days residence through saturated organic layers, compared with 15-40 years travel time predicted in the groundwater study, was sufficient to ensure that effectively no nitrogen sourced from the WWTP would enter the harbour. The study was accepted by the regulator and a 35-year consent was granted.

KEYWORDS

Nitrogen, fate and transport, soils, peat, irrigation, denitrification, WWTP, stochastic model

PRESENTER PROFILE

Dr Jim Cooke is a water quality specialist with 40 years' experience in environmental science, principally directed towards the management of receiving waters. He has specialised in non-point (diffuse) pollution both in respect to rural and urban land uses, as well as incorporating science knowledge within National and Regional Planning processes.

1 INTRODUCTION

The Omaha WWTP (operated by Watercare Services Ltd – WSL) is located at Jones Rd, Omaha Flats, approximately 70 km north of Auckland CBD. The plant treats wastewater from the townships of Omaha, Point Wells and Matakana. Resource consents for the plant expired in 2015 and the application for renewal included provision for a near doubling in wastewater volumes from the current 157,000 m^3/y to 300,000 m^3/y within 15-20 years due to projected population increases.

Wastewater treatment at Omaha comprises an aerated lagoon, oxidation pond, storage dam, tertiary filters and UV disinfection. The treated wastewater is irrigated on to a 7.6 ha block of mature eucalypts and a 5.5 ha block of native plants (mainly manuka and kanuka) at Jones Rd, adjacent to the WWTP. In addition, treated wastewater is piped across the Broadlands Drive Causeway to the southern half of the Omaha Beach Golf Club (OBGC) on the Mangatawhiri Spit and irrigated to a 5.7 ha block of mainly fairways and 0.6 ha of dunes. The WWTP and irrigation areas are described in more detail in Stuart et al. (2017).

The Jones Rd and OBGC irrigation blocks are located on opposite sides of Whangateau Harbour (Figure 1) which is an area of high ecological significance. Consultation at the start of the consent process showed that stakeholders were concerned about the potential for increased nitrogen (N) loads from irrigated wastewater to leach through to groundwater and cause a deterioration in water quality. Because of this concern, WSL commissioned a study of the fate of transport of the irrigated N. In this paper we report on the results of this study, which was one of a number of studies informing the Assessment of Ecological Effects. Other interrelated studies included hydrogeology and groundwater modelling, hydrodynamic modelling of Whangateau harbour, emerging organic contaminants, and ecological studies both of the harbour biota and Kahikatea wetland between the OBGC and the harbour (Figure 1). Results from the hydrogeology and groundwater modelling studies were presented at the 2017 Water NZ conference (Stuart et al., 2017) and we have utilised some of these findings (particularly groundwater flow directions and travel times) in this study.

Figure 1: Omaha study location showing Whangateau Harbour, the WWTP, the Jones Rd eucalyptus and native irrigation blocks, the OBGC (southern end), the dunes block, and the Kahikatea forest.

Regardless of whether irrigation of treated wastewater is to the OBGC or Jones Rd sites the processes that lead to either loss, immobilisation or transport (to Whangateau Harbour in this instance) are complex. A simplified N cycle for such a system, in which there are pathways for loss as well as transformation, is given in Figure 2.

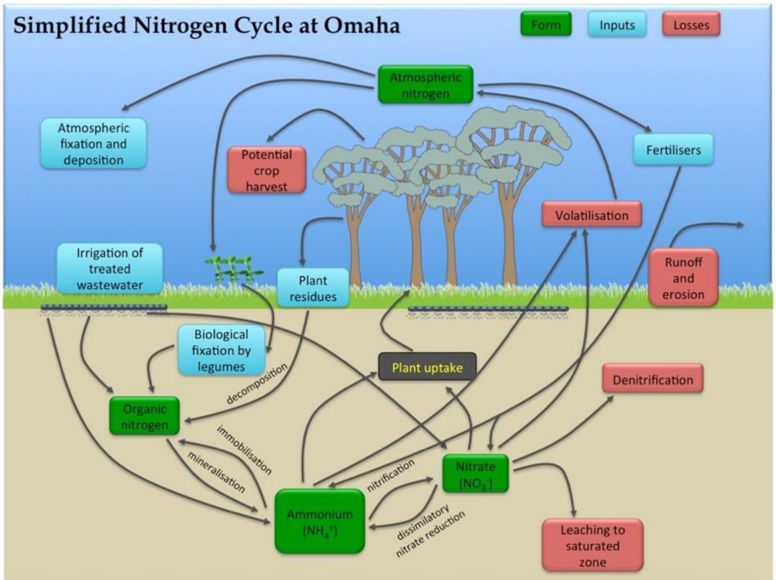


Figure 2: Simplified nitrogen cycle for irrigated wastewater at Omaha

Figure 2 shows that potential pathways whereby irrigated N can be <u>lost</u> from the system include crop harvest, immobilisation, ammonia volatilisation, runoff/erosion, denitrification and leaching. For this study we concentrate on denitrification and leaching as the only two vectors for loss from the 'irrigation site'. This is because (i) there is either a long term (as is the case with the eucalypts) or no crop harvest (even the golf course clippings are left to decompose and not removed), (ii) for ammonia volatilisation to be significant soil pH needs to be alkaline, whereas at both the OBGC and Jones Rd pH is weakly acid, and (iii) the topography is generally flat and there is no visible indication of tunnel or gully erosion. Denitrification is the only process whereby N is completely removed from land and aquatic ecosystems through the end products being nitrogenous gases that are lost to the atmosphere. It generally proceeds through some combination of the following

NO_3^-	→	NO_2^-	→	NO(g)	→	N_2O(g)	→	N_2(g)
nitrate		nitrite		nitric oxide		nitrous oxide		nitrogen

Denitrification is mediated by facultative anaerobic bacteria (which prefer oxygen if present, but which are capable of switching to anaerobic respiration if oxygen is absent), and many bacterial groups are so classified. Denitrifying bacteria require low oxygen and a source of available carbon for energy. Such conditions were thought to exist within the irrigation leachate zone (DSL, 2008), particularly the Kahikatea Forest/wetland between the golf course and harbour, the peat deposits beneath the eucalypts, and the intertidal mud flats. As part of an initial phase 1 investigation, we undertook denitrification enzyme activity (DEA) assays of soil cores for 15 sites both within the irrigation areas and at other sites at the edge of the Kahikatea forest/wetland and within the intertidal mud flats chosen to be representative of the conceptual flow path. DEA provides an estimate of the potential for denitrifying organisms at each site to denitrify nitrate to nitrogenous gases. This work (SEL, 2015) showed moderately high rates of DEA on the Jones Road site, particularly in the surface layers but with measurable activity throughout 1-2 m length cores. In contrast there were high rates in the surface (A) horizon of the golf course soils, but no or negligible activity below this depth. DEA within the Kahikatea Forest soils and the mud flats was quite variable.

The DEA results only enabled us to infer denitrification rates when nitrate was present, oxygen was absent, and diffusion was non-limiting. To get a more realistic estimate of actual denitrification we needed to use an *in-situ* assay. We also needed to understand the available carbon supply, which is necessary to sustain denitrification. Further, to understand the fate of irrigated N, which is dominantly in ammonium form, we needed to measure nitrification rate (Figure 2). Finally, to model N transport to the saturated zone, we needed a chemical measure of nitrate and ammonium in the soils. These measurements were incorporated in the second phase of the study, which we discuss in this paper. At the time of starting Phase 2 the hydraulic flow paths were not known, so we concentrated our measurements beneath the irrigation sites.

It may be noted in Figure 2 that there are additional processes that influence N loss from the irrigated sites. Many of these processes occur at such a slow rate that we were unable to make meaningful measurements of their rate. However, plant uptake was important to stakeholders and we commissioned Scion (Simeon Smaill) and Turf and Landcare Science Pty Ltd (Keith McAuliffe) to estimate N uptake by the eucalypts, and immobilisation on the golf course, respectively.

In this paper we combine our understanding of the "fate" of irrigated N gleaned from biochemical assays and chemical measures, with its "transport" thorough firstly the unsaturated soil and secondly the saturated (groundwater) zone. We utilised the findings of a parallel workstream on groundwater modelling (Stuart et al., 2017) to provide parameter values for our N transport model, as well as giving us information on flow paths and dimensions and locations of peat deposits, which provided added confidence to our assessment. The model we developed was risk-based and high-level, but we consider it was fit for purpose, and representative of the current state of knowledge. It was useful as a high-level screening tool for illustrating the effects of management changes to stakeholders, and to provide estimates of N renovation of irrigated wastewater and the likely impacts on Whangateau Harbour.

2 METHODS

2.1 FATE OF NITROGEN

2.1.1 SOIL SAMPLING

Soil samples were collected in two separate field trips in 2015; one in unsaturated soils by ourselves, and the 2nd within the saturated zone by PDP in association with bore hole drilling. Five sites were sampled in or around the OBGC area: two on the golf course - GOLF1 and GOLF2; and three in the adjacent Kahikatea forest/wetland - KAH1, KAH3, KAH5. Four sites were sampled from the Jones Rd area: two in the Eucalypt plantation - EUC1 and EUC2; and two in the native vegetation area - NAT1 and NAT2 (Figure 3). All 9 sites were sampled by hand auger, from which soil was collected at 3 depths; sub-surface (designated -1); mid-range (-2) and at interface with saturated zone (-3). Notes were taken during sampling of soil type (e.g. peat, clay, sand) and moisture content (e.g. dry, moist, wet) as well as redox potential (Eh) in the saturated zone sample.

Soil samples were collected 30 minutes after an irrigation event was completed. The volume of Omaha WWTP wastewater irrigated varied for each site. Samples were placed in plastic bags, labelled, sealed and bagged again before being placed in a chilly bin. *In-situ* DEA experiments and KCl/$HgCl_2$ extractions for NH_4/NO_3 analysis were carried out as soon as was practical after collection of each set of samples (see section 2.2). This was generally within 5 min from all sites except the Kahikatea forest soils, which due to logistical issues of not being able to transport gas cylinders, was delayed for up to 15 min. Remaining soil samples were transported on ice back to the laboratory for denitrification enzyme activity (DEA), readily mineralizable carbon (RMC), and nitrification rate experiments.

PDP undertook bore drilling at 13 sites (Figure 3). Soil samples were taken at the saturated zone at each site and nitrate/ammonia extractions carried out on site using a SEL supplied standard operating procedure. Samples were transported on ice back to the laboratory for measurement of DEA, and dissolved organic carbon (DOC) as below.

2.1.2 ASSAYS AND ANALYSES

The methods for the biochemical assays DEA, *in situ* denitrification activity, RMC, and short-term nitrification activity (SNA) are detailed in Streamlined Environmental (2015). The chemical methods for nitrate and ammonia, DOC, and loss on ignition are also specified in the same reference. It is important to note that *in situ* denitrification assays and 2M KCl (+40 ppm $HgCl_2$) extractions for nitrate and ammonia were carried out in the field following soil sampling.

2.1.3 N LOAD CALCULATIONS

A calculation of the load of N applied by irrigation was carried out using TN monitoring data (g/m^3) and daily wastewater discharge volumes (m^3) for 2014/15. TN data (n=23; approximately fortnightly) were provided for the time period of 19/08/14 and 23/06/15. Daily discharge volumes to Omaha Beach Golf Course and Jones Rd areas were provided for 1/07/14 to 30/06/15. Discharge volumes to each site and TN concentrations differ between summer (defined as October to April) and winter (defined as May to September). Therefore, *total* discharge volumes (m^3) to each site along with *average* TN of the wastewater were calculated over the summer and winter periods separately.

Figure 3: Site locations for soil collection. Golf course sites (top); Jones Rd sites (bottom). Omaha WWTP highlighted at Jones Rd site. Green symbols are bore sites and red symbols are soil sampling sites.

2.2 TRANSPORT OF NITROGEN

2.2.1 MODELLING OBJECTIVES AND OVERVIEW

We developed a numerical model to provide for screening level analysis of likely N flow paths of Omaha WWTP land irrigation water. The model is considered “high level”, with recognised uncertainties associated with both the timing and magnitude of projected N concentrations and loads in the shallow subsurface. That being said, the model is an appropriate approach designed to provide important insight into the relative distribution of load to various biophysical compartments in the system (utilising the results of biochemical assays and chemical analyses as well as order of magnitude estimates of N concentrations in the subsurface and loads entering the deeper saturated groundwater zone.

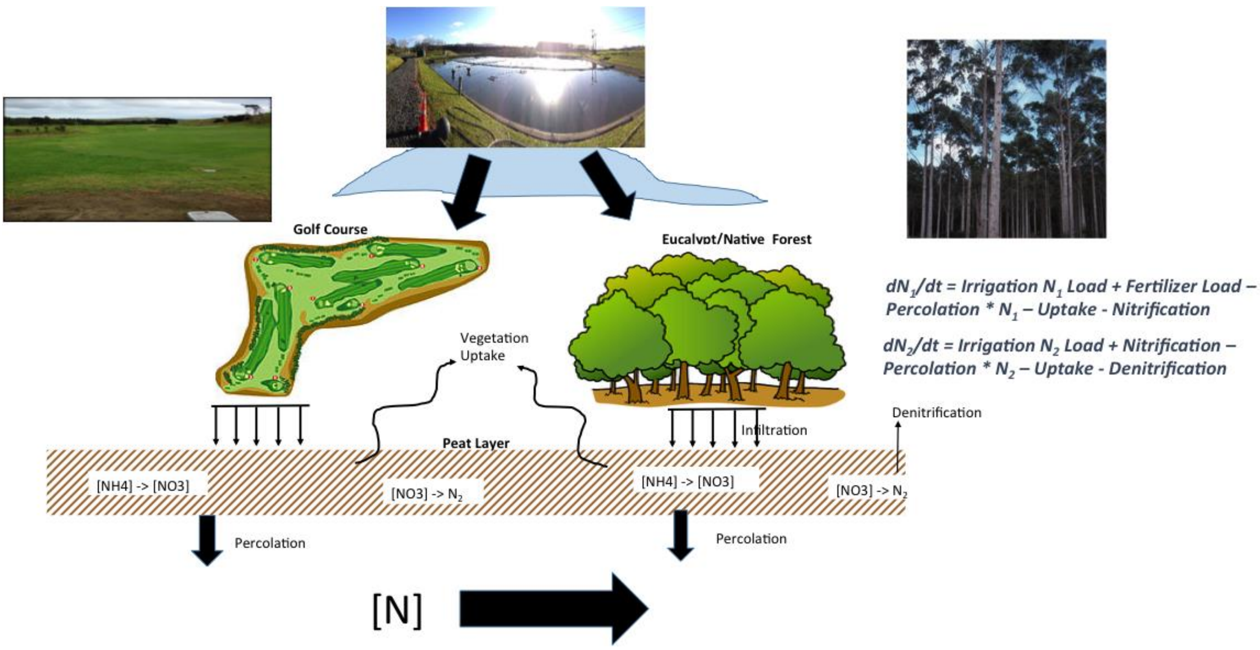
The model simulates the application of Omaha WWTP treated wastewater, as irrigation, to multiple land treatment zones. The model domain focuses on the shallow unsaturated subsurface associated with the two primary irrigation areas, as described above: Jones Road (divided into separate "Eucalypt" and "Native Block" zones) and the OBGC (divided into separate "Fairways" and "Dunes" zones). Nitrogen loads resulting from these practices are simulated in the model on a daily timestep. Nitrogen fate in the subsurface is simulated as a function of water movement through the unsaturated zone (infiltration and percolation), vegetation uptake, and nitrification-denitrification processes occurring in the subsurface. Model output is in the form of daily ammonia-N, nitrate-N, and total-N concentrations in unsaturated zone pore water and total-N loads to the deeper (saturated zone) subsurface. Additionally, a simplified estimate of final saturated zone irrigation water N concentrations and loads, prior to discharge to receiving waters and after a prolonged residence in the saturated zone, is provided by the model as a function of an assumed first order rate and prescribed residence times.

In recognition of the relatively high uncertainty levels associated with many of the model inputs, model calculations are performed within a probabilistic framework with model inputs defined by a distribution of values rather than single values. During each model simulation, input distributions are sampled randomly (stochastically) over 1000 sampling iterations. Final outputs are provided as cumulative probability distributions that, most directly, reflect the levels of input parameter "consensus" associated with output thresholds. Less directly, the output distributions reflect levels of "risk" associated with the concentration or load thresholds. This approach is deemed appropriate for this study as a means of demonstrating modelling uncertainty, capturing the ranges of possible outcomes, and better supporting decision-making.

2.2.2 MODEL CONSTRUCTION AND PARAMETERIZATION

The Omaha WWTP N model (Figure 4) was constructed in Microsoft Excel, with the stochastic add-in @Risk (Palisades Incorporated). It simulates an extended continuous climate period (1967 – 2014) on a daily timestep. Four different land application sites were included and simulated separately in the model: Golf Course, Eucalypt Forest, Native Forest blocks and "Dunes". Daily irrigation rates were prescribed based on reported data for the 2010 - 2014 irrigation seasons, with repeating sequences for each of the remaining years in the 48-year simulation period. Additionally, a steady fertilizer application rate (100 kg ha^{-1} $year^{-1}$) was assumed for the Golf Course zone.

a) Nitrogen accounting model



b) Soil Moisture Model

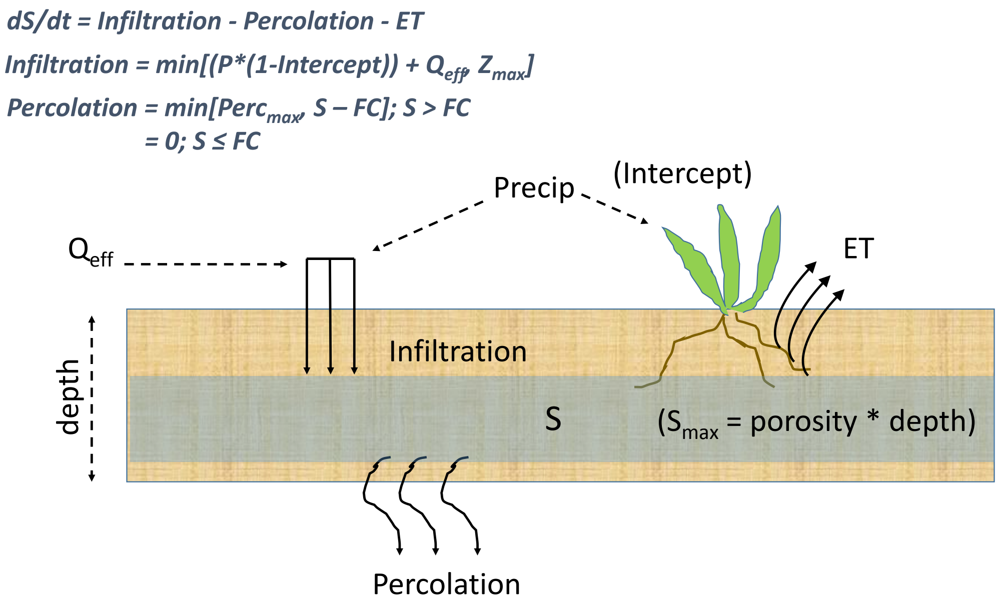


Figure 4: Omaha Beach WWTP Irrigation Model Components (a) Nitrogen accounting and (b) Soil Moisture.

Nitrogen loads and surface water enter the subsurface unsaturated zone in the model via infiltration. Surface water infiltration is calculated as the sum of daily precipitation (P), minus interception, and prescribed irrigation water application, subject to a maximum infiltration rate. Upon infiltration, surface water enters the soil moisture pool held in the delineated subsurface zone. Daily soil moisture volumes (S) are calculated as a function of the start of day antecedent moisture, daily infiltration water, percolation, and evapotranspiration (ET), using a simple water balance (Figure 4b). Percolation is calculated as the difference between the soil field capacity (FC) and the total soil moisture available to percolate if soil moisture is greater than field capacity. If the soil moisture is not greater than the field capacity threshold, percolation is assumed to be zero. Evapotranspiration is calculated in the model as a function of prescribed daily potential evapotranspiration (PET) rates, available soil moisture, and a specified "wilting point" for the soil zone. If the available soil moisture (antecedent S + P) is below the prescribed wilting point, then ET = 0 for that day. Otherwise ET equals either the available soil moisture above the wilting point or the prescribed daily PET, whichever is less. Daily PET rates in the model were estimated using the Penman method. Lastly, a user-defined maximum holding capacity (S_{max}) of the modelled soil zone places a cap on the total pore water volume. If the soil moisture reaches S_{max}, then no further infiltration occurs until space is created (via ET and percolation). In this case, or in the case of available surface water exceeding the maximum infiltration rate, the model calculates a surface runoff volume that does not infiltrate to the subsurface.

Coupled with the soil moisture calculations, N mass balance calculations are performed in the model for the same designated subsurface zone (Figure 4a). Prescribed monthly N concentrations (NH_4-N and NO_3-N) associated with the irrigation water provide the majority of the mass loading to the subsurface. Additionally, for the Golf Course zone only, a steady monthly fertilizer load (NH_4-N) is assumed. One hundred percent of the fertilizer load is assumed to reach the subsurface. Irrigation water N load is transported to the subsurface via infiltration. Therefore, if the subsurface is saturated and irrigation water runoff occurs, not all of the load reaches the modelled zone (proportional to the amount of total irrigation water that infiltrates). Within the modelled subsurface zone, pore water N concentrations are calculated as a function of: soil moisture volume, vegetation uptake, nitrification – denitrification processes, and mass leaching to the deeper subsurface (via percolation). Vegetation uptake is assumed to be a zero-order process in the model, while nitrification and denitrification are modelled as first order processes.

The above algorithms apply to treated wastewater irrigated on the eucalypts, native block, and fairway areas of the OBGC. However, during winter, or extreme wet weather at other times when OBGC do not accept fairway watering, a portion of the wastewater load is irrigated on the dunes area to the east of the golf course. These dunes are largely unvegetated sand. Their free-draining oxic nature and absence of organic matter lead us to conclude there will be zero denitrification in this dune area. This is borne out by nitrate concentrations > 2 mg N/L found in the newly installed (2015) bores by PDP. Whilst most of the groundwater beneath the dune areas flows eastwards towards the ocean, some flows westward towards the harbour. Further, the groundwater divides beneath the dunes shifts as a function of groundwater mounding (PDP, 2015). Therefore, for the purposes of modelling N loss and transport from treated wastewater irrigated on the dunes area we have assumed:

1. There is no attenuation within the unsaturated zone. i.e. all N irrigated N leaches to the groundwater table,

2. All N reaching the groundwater table beneath the dunes is nitrified (i.e. as NO_3^-).

3. The wastewater-derived N in the portion of groundwater beneath the dunes flowing towards the harbour (as defined by PDP (2015)) is subject to attenuation in the active saturated zones as discussed below.

The fate of leached N in the deeper saturated zone (groundwater) is approximated with a simple first order decay model. Saturated zone N is assumed to be subject to prolonged residence time (estimated by PDP based on their groundwater modelling) and to additional losses via denitrification and sediment immobilisation in biologically active peat and marsh subsurface layers. The following equation is used to estimate final N concentrations in the saturated layer (prior to discharge to receiving waters):

$$C_f = C_0 e^{-k t_r}$$

Equation 1

where C_f = final N concentration after passing through saturated zones, C_0 = initial N concentration of leached irrigation water, k = assumed first order denitrification (NO_3) or sorption/immobilization (NH_4) rate constant, and t_r = assumed residence time in "active" saturated zone. This equation is solved for each timestep in the model simulation period and each stochastic iteration. Note that mixing with other groundwater and/or N loads is neglected for this exercise, as only the irrigation water "parcels" are tracked through the saturated zones.

The water and mass balance calculations are performed in the model for a range of input values over 1000 stochastic iterations. Uniform distributions of values were assumed for all stochastic inputs (ranges provided in Table 1). Trial simulations showed that final results are insensitive to additional iterations (beyond 1000). "Latin Hypercube" was selected as the stochastic sampling method, which is known to decrease the number of required iterations in such models.

Table 1: Summary of Omaha Beach Nitrogen Model Inputs

Parameter	Values	Stochastic?	Source
Meteorology and Irrigation:			
daily precipitation	variable (1967 – 2014)	No	climate stations (PDP, 2015[1])
daily ET	variable ("")	No	Penman PET * Crop Factor (PDP, 2015)
daily irrigation	variable (2014 – 2015)	No	WSL data
application area	5.7 ha (GC); 7.6 ha (EUC); 5.5 ha (NAT); 0.62 ha (DUN)	No	WSL data
wastewater N concentrations	1 – 28 g-N m^{-3}	No	WSL data
interception fraction	0 (GC, DUN); 0.1 – 0.5 (EUC, NAT)	Yes (EUC,NAT)	literature (see SEL, 2015)
fertilizer application	100 kg-N ha^{-1} yr^{-1} (GC); 0 (EUC, NAT, DUN)	No	Turf and Landcare, 2015
Hydrogeology:			
active layer depth	75 - 225 mm (GC); 500 - 1500 mm (EUC); 250 - 750 mm (NAT)	yes	PDP, 2015 (± 50%)
porosity	0.4 – 0.6 (GC, DUN); 0.7 – 0.9 (EUC, NAT)	yes	PDP, 2015 (± 50%)
max infiltration rate	200 – 5000 mm d^{-1} (GC); 20 – 600 mm d^{-1} (EUC, NAT)	yes	field testing (PDP, 2015)
max percolation rate	100 – 300 mm d^{-1} (GC, DUN); 10 – 30 mm d^{-1} (EUC, NAT)	yes	PDP, 2015 (± 50%)

Parameter	Values	Stochastic?	Source
field capacity	15% of depth (GC, DUN); 70% of depth (EUC, NAT)	yes (via depth)	PDP, 2015
wilting point	5% of depth (GC,DUN); 30% of depth (EUC, NAT)	yes (via depth)	PDP, 2015
Biokinetics:			
vegetation uptake rate	0 – 0.04 g-N m^2 d^{-1} (GC); 0 – 0.002 g-N m^2 d^{-1} (EUC, NAT); 0 (DUN)	yes	Turf and Landcare, 2015 (GC); Scion, 2015 (EUC) ; Franklin, 2015 (NAT) -see SEL (2015)
denitrification rate	0.07 – 0.59 d^{-1} (GC); 0.04 – 0.12 d^{-1} (EUC); 0 – 0.07 d^{-1} (NAT) 0 (DUN)	yes	SEL field assays
nitrification rate	0 – 0.09 d^{-1} (GC) 0 – 0.12 d^{-1} (EUC) 0 – 0.13 d^{-1} (NAT) 0 (DUN)[1]	yes	SEL assays
soil temperatures	10 – 20 °C	no	NIWA CliFlow database
ammonia uptake preference factor	1.0	no	assumed
Saturated Zone Kinetics:			
active saturated zone residence time	1 – 2.5 years (GC, DUN) 1 – 4 years (EUC, NAT)	yes	PDP, 2015
active saturated zone denitrification rate	0.005 – 0.01 d^{-1} (GC, DUN) 0.005 – 0.1 d^{-1} (EUC, NAT)	yes	SEL field assays[2]
active saturated zone NH_4 sorption rates	0.009 – 0.018 (GC, EUC, NAT, DUN)	yes	literature (see SEL, 2015))

3 RESULTS AND DISCUSSION

3.1 FATE OF NITROGEN

3.1.1 ASSAYS AND CHEMICAL MEASUREMENTS

While assays and chemical measures were made on soils from OBGC, and the within the Kahikatea forest on the eastern side of Whangateau Harbour, the groundwater modelling

[1] The decision to include the dunes was made after our field and laboratory analyses were completed. The newly installed bores (July 2015) showed high nitrate concentrations and low ammonium concentrations indicating nitrification occurring on the irrigated dunes. For the purpose of this model we assumed a fully nitrified effluent reached the saturated zone.

[2] Unable to estimate rate constants for deeper saturated zones due to absence of measurements. We used the lower range of values from the shallow saturated zones to be conservative.

(PDP,2015) showed that treated wastewater irrigated on to the golf course travelled beneath the Kahikatea forest to emerge in in the intertidal mudflats. Therefore, the results of rate measurements with the Kahikatea forest were not relevant to the ultimate transport of N to Whangateau Harbour and are not presented here. Similarly, the flow path of treated wastewater irrigated to the native block is complex, and the results of the assays, whilst scientifically interesting, are also complex. Therefore, because of space limitations we only present the results of assays from OBGC and the Jones Rd eucalypt forest graphically, though the results obtained from the native block are briefly discussed.

A. Jones Rd Eucalypts block

Denitrifying enzyme activity (DEA) was relatively high in the surface sites, but activity dropped off rapidly with depth (Figure 5). In contrast, *in-situ* denitrification rates were much more uniform with depth, with rates similar to subsurface DEA rates. The *in situ* denitrification rates were sufficient to account for all nitrate losses in the profile. Even though the soil coring started within 30 minutes of irrigation ceasing, nitrate concentrations were relatively low[3] with a maximum concentration of 2.9 µg NO_3-N/g recorded in the surface EUC 1 sample. Ammonium-N was very much higher (8-10 µg N/g) and relatively evenly distributed throughout the 2-2.5 m core. This is likely due to a combination of mineralisation of organic N in the peat substrate and immobilization of NH_4-N from the treated wastewater. Readily mineralisable carbon (RMC) levels were similarly high and evenly distributed. RMC levels were more than sufficient to sustain measured levels of denitrification. The loss of ignition concentrations confirmed the large reservoirs of carbon expected in a peat soil.

Short term nitrification results were ambiguous with moderately high levels down to 600 mm in the EUC 1 sample, but no activity at depths >600 mm in the EUC 2 sample. Zero activity was also recorded at the deepest depth in both samples. Thus, although there appears to be ample ammonium, and aerobic conditions (at time of sampling) it appears that in this high carbon environment, conditions are not conducive for nitrifying bacteria to compete with heterotrophs (that use carbon for energy).

B Omaha Beach Golf Course

High DEA rates were measured in the surface layer of both cores but only trace activity in subsurface layers (Figure 6). In-situ denitrification followed a similar pattern. RMC was also higher in the surface layers, consistent with the strong turf formation beneath the fairways, but dropped rapidly below the turf zone where the coring showed just mineral sand. The GOLF 1 core (Figure 3) appeared to have more organic matter (RMC, LOI) with depth but levels were minor compared with the eucalypt sites. Only trace amounts of nitrate were extracted at all depths from both cores. Ammonium levels in the GOLF 1 core were significantly higher than GOLF 2 and increased with depth. The high levels of ammonium in subsurface GOLF 1 core were matched by high nitrification activity in the 600-1300mm range. An increase in nitrification activity was also noted in the GOLF 2 core at the same depths but not to the same extent.

The overall picture emerging from the Omaha Golf course irrigation site is one of good organic matter build up in the surface layer with moderate DEA (i.e. denitrification will occur if nitrate is present). Our turf expert (Turf and Landcare Science, 2015) estimated high rate of N immobilisation (turf uptake and decomposition) in the surface 'turf' zone and this would appear to be the more important process than denitrification. Below the turf zone there is little organic matter, and interestingly we detected significant nitrification in this zone, but only negligible nitrate. This indicates that nitrifying bacteria have

[3] All expressed as µg N/g soil dry weight basis

accumulated in this zone, which will nitrify any free ammonium present if conditions are conducive. Given there was elevated ammonium in these layers (in the GOLF 1 core anyway) we would have expected to extract more nitrate than we did. A likely explanation is that nitrate generated by nitrifying bacteria may have been leached below the zone of activity in this free-flowing sandy soil by the previous irrigation event. Despite this anomaly there is evidence that nitrification is taking place below the turf zone and with low organic matter there is little potential for it to be lost via denitrification in the immediate leaching zone.

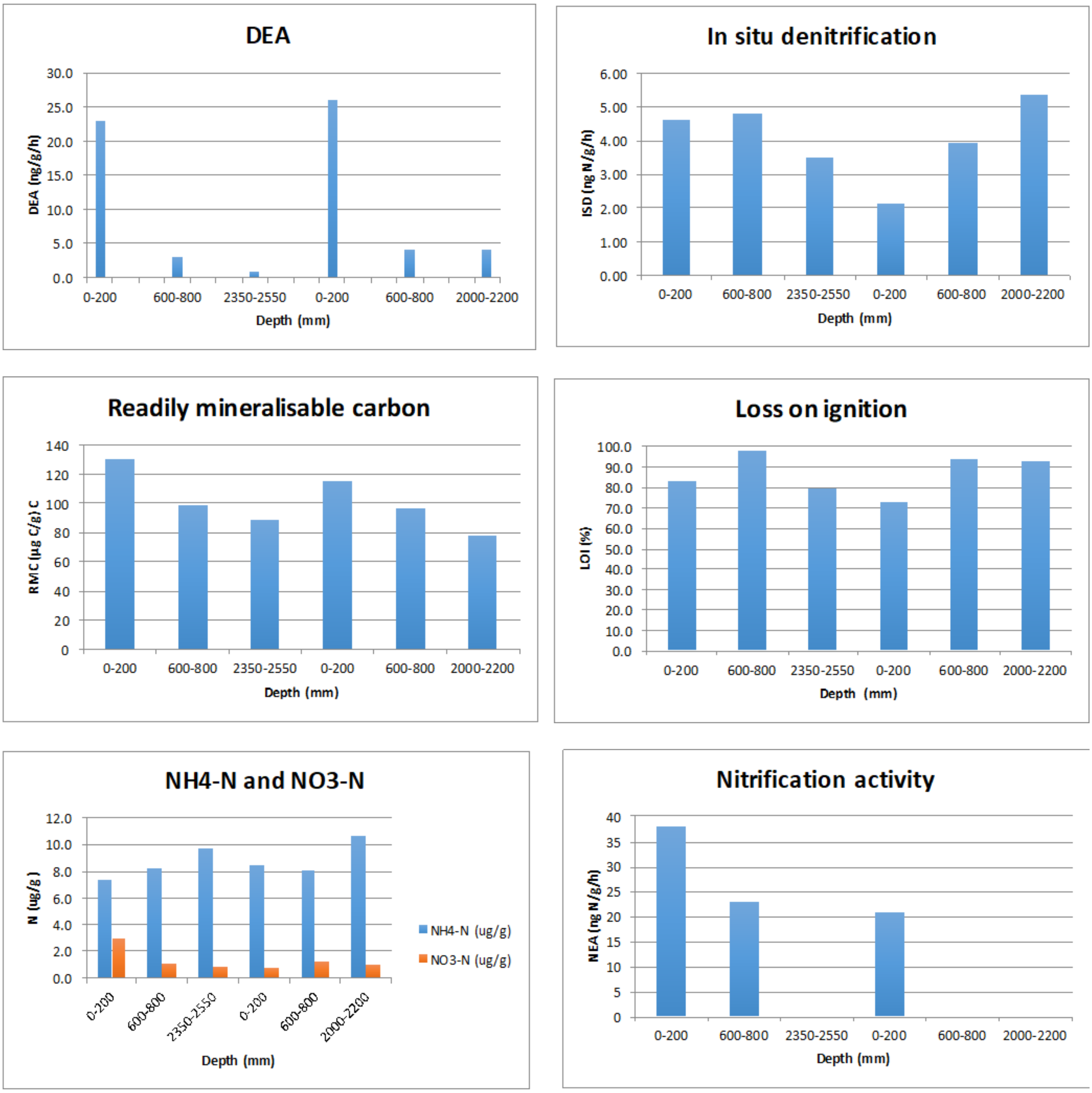


Figure 5: Results of assays and chemical measurements on soil samples collected from the Eucalyptus Forest (EUC1 On LHS and EUC2 on RHS of each graph (see Figure 3).

The main driver for sustainability of denitrification is available carbon. Denitrifiers are heterotrophs and need carbon as a source of energy. Since a landmark paper (Burford and Bremner, 1975) reported a relationship between the amount of organic matter in soil and the rate of denitrification, many researchers have used readily mineralizable carbon as a predictor of denitrification.

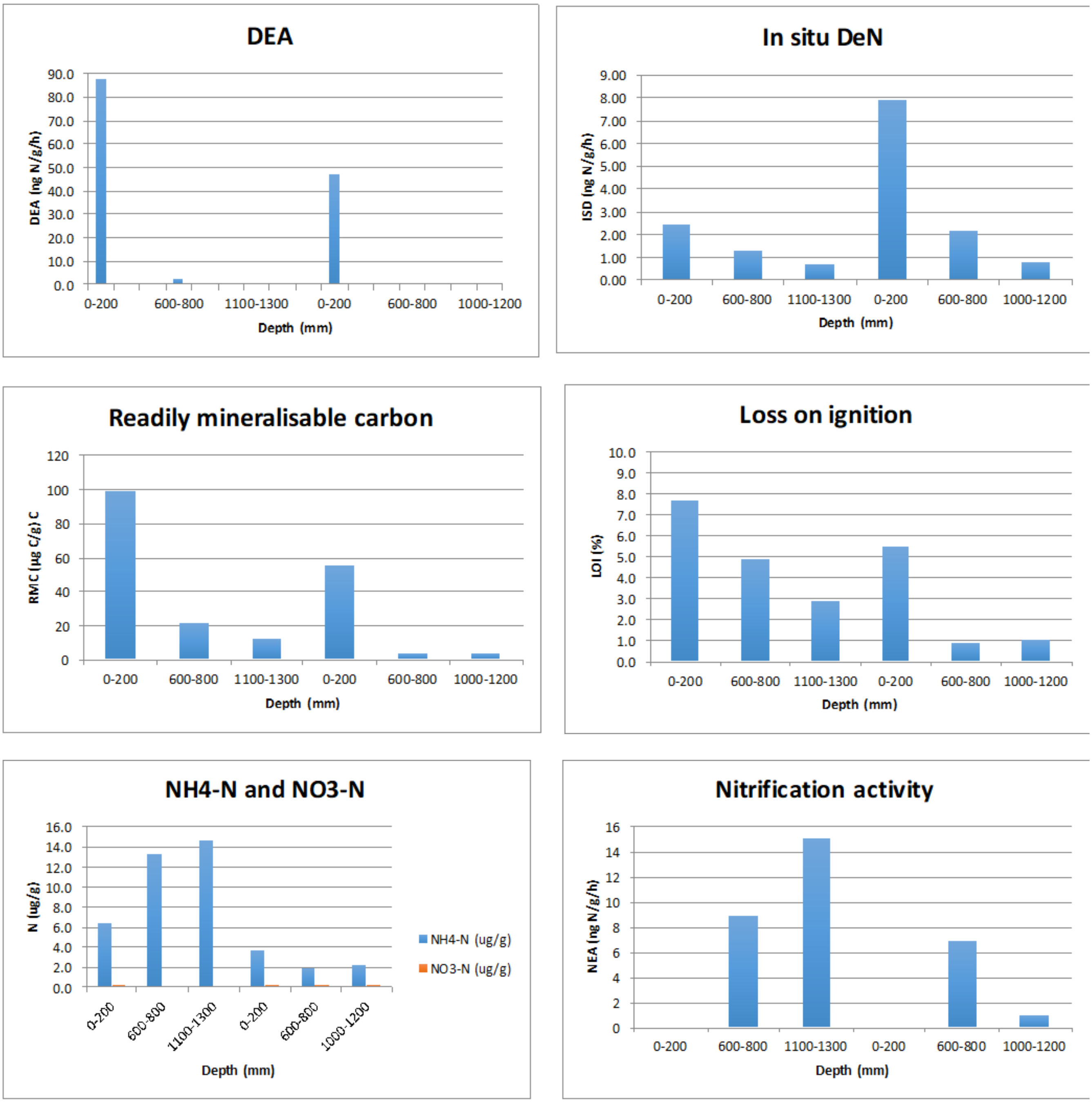


Figure 6: Results of assays and chemical measurements on soil samples collected from the Eucalyptus Forest (EUC1 On LHS and EUC2 on RHS of each graph (see Figure 3).

We found some clear relationships between RMC and DEA for both the OBGC and Eucalypt sites (r^2=0.98 and 0.73, respectively). However, these tended to be driven by a cluster of high points (mainly surface horizons) and a cluster of low points (subsoils) with little in between, rather than a continuum of data points.

C. Saturated zone assays and chemistry

Results were generally consistent with those from the unsaturated zone. Ammonium-N concentrations were variable with some high concentrations being recorded (e.g. 48.3 µg/g at MS12 on the northern part of OBGC, an area that is not irrigated with Omaha wastewater, whereas the neighbouring MS11 was only 3.3 µg/g). These could not be matched to equivalent unsaturated samples because they were taken at different sites, including some in the dunes area and in the suburban area to the east of the dunes (Figure 3). Following completion of our field and laboratory work, a greater emphasis was placed on dunes and eastward flow paths, because the bores installed at these sites showed elevated nitrate concentrations (see notes on modelling assumptions Section 2.2.2).

D. Summary of uptake and immobilisation estimates

Eucalyptus forest

Significant points from the Scion report (Smaill, 2015) are summarised below.

The Eucalyptus forest at Jones Road is a 10 ha stand (7.6 ha irrigated) currently comprising approximately 90% *Eucalyptus botryoides* and 10% *Acacia melanoxylon*. *A. melanoxylon* were either planted or have otherwise became established in rows the initial planting. The remaining *E. botryoides* are around 27 years old. It can be assumed with high confidence that foliar biomass is no longer increasing so demand for N to support canopy expansion is unlikely.

Smaill (2015) estimated that current N uptake will be in the range 0-10 kg N/ha/y. *A. melanoxylon* is an N-fixing legume and the rate of fixation increases with age. Smaill reasoned that when the N-fixing impact of *A. melanoxylon* is considered, the N uptake by the irrigated *Eucalyptus* forest may be close to 0.

Golf Course

Significant points from the Turf & Landcare Science Pty Ltd (T&LC Science) (2015) are summarised below.

Fertiliser N is currently applied to the fairways at a rate of 100 kg N/ha/year in the form of ammonium sulphate. A total of 7.5 ha is currently irrigated. Typically, 100-200 m^3 is irrigated in winter (May to September) and 1000m^3 in summer (October to April). There is no legume content, so the only other N input is assumed to be wastewater irrigation.

The information provided also indicates that an average of 128kg N/ha/yr is applied to the fairways via the irrigated wastewater. Therefore, T&LC Science (2015) estimated that ~ 228kg N/ha/yr is applied to the fairways and tees.

T&LC Science (2015) noted that from an agronomic perspective, N removal occurred via two processes: soil immobilisation and leaching through the profile. Soil organic matter testing indicated that 2100 kg N/ha is stored in the soil profile, much of which is likely to have accumulated over the past 12 years of irrigating. This indicates that up to 175 kg N/ha/yr could have accumulated in the soil profile, with an estimated 50 kg/ha/yr of N being leached. He noted that because the N removal was related to good turf management, the immobilisation rate was probably sustainable.

3.2 TRANSPORT OF NITROGEN

Mass balance summaries of model projected N fate in the unsaturated zone for the golf Course and Eucalypt sites are provided in Table 2. Note that these particular results were calculated deterministically (rather than stochastically) using median values of the stochastic parameters. They are intended to demonstrate the relative distribution of N pathways, as predicted by the model. The highest N loads, associated with the irrigated areas (non-Dunes) both applied and leached, were associated with the golf course fairway irrigation. The percentage of mass lost to a combination of vegetation uptake/immobilisation and denitrification is also lowest for the golf course site (Table 2) compared to the eucalypt and native bush sites. As a consequence, average annual TN concentrations in the leached irrigation water are nearly ten times higher for the golf course compared to the other two zones. This is despite combined uptake/immobilisation and denitrification rate constants are being slightly higher for the golf course. This is explained by the residence times in the golf course unsaturated zone, which are significantly lower than those in the analogous eucalypt and native zones, due to a shallower depth and higher percolation rates. For both the eucalypt and native zones, denitrification, under assumed median value conditions, removes the vast majority (83 – 90%) of applied N prior to leaching. For the golf course, again assuming median rate values, a lesser, but still significant, amount of N (65%) is lost via a combination of vegetation uptake/immobilisation and denitrification. The highest N loads lost to leaching across all application zones are those associated with the dunes application, where N is assumed to be fully conservative (131 kg/N applied, 131 kg/N leached).

Stochastic modelling results are summarized in Figure 7. Included here are summaries of leached N concentrations and loads leaving the modelled unsaturated zones; and final N concentrations after a prolonged residence in the biologically active saturated zone. Results, in the form of cumulative probability distributions, provide insight on the range of uncertainty in predictions as a result of the uncertainty associated with the input parameters. Figure 7 (top) shows the range of predictions of annual leached N concentrations, averaged over the full simulation period, leaving the unsaturated zone and entering the saturated zone, for each irrigation area. Modelled N concentrations range from approximately 3 to 25 g m^{-3} for the golf course, approximately 1 to 16 g m^{-3} for the native zone, and approximately 1 to 6 g m^{-3} for the eucalypt zone. For the fully conservative dunes application, modelled N concentrations range from approximately 17 to 25 g m^{-3}. Figure 7 (middle) presents the corresponding loads of leached N leaving the unsaturated zone (concentration x flow). Modelled N loads leached from the unsaturated zone range from 200 to 700 kg yr^{-1} for the golf course and approximately 1 to 50 kg yr^{-1} for the two forested areas, and 500 to 550 kg yr^{-1} for the dunes. Lastly, Figure 7 (bottom) shows predicted final irrigation water N concentrations after moving through the biologically active saturated zone and assuming first order losses (as described above). In other words, these are the projected N concentrations for groundwater leaving the subsurface and entering receiving waters. Projected groundwater N concentrations associated with the two forest zone applications are projected at less than 0.02 g m^{-3} for the entire distributions. Modelled groundwater N concentrations associated with the golf course irrigation range from 0 to 0.2 g m^{-3}, while those associated with the dunes application range from 0 to 0.4 g m^{-3}.

These results highlight the disparity in N removal efficiencies between the golf course and the two forested zones. Leached loads and concentrations are projected to be much higher for the golf course compared to the other two. As described above, this is due to a combination of higher applied loads and shorter residence times. Also evident from Figure 9 is the significant N removal projected for the deeper, active, saturated zones. Assumed residence times in these zones range from 1 to 4 years. Even at relatively low assumed rates of denitrification and immobilisation, only trace levels (<0.01 mg N/L from Jones Rd side) of N are projected for the water discharging from these zones.

The above simulations were performed on the "baseline" scenario, in which ~164,000 m^3 of treated wastewater was irrigated annually. Because of projected population increases a scheme was designed (Scenario E, PDP, 2015) whereby up to 300,000 m^3/y could be irrigated without surface flooding. This was achieved largely by increasing the area of irrigation on the native block and optimising irrigation scheduling between the blocks according to season. We carried out further simulations using the predicted volumes. Because of the optimised scheduling, the predicted distribution of N concentration and loads for Scenario E were all equal to or less than the baseline scenario with the exception of the dunes block. For the dunes, significantly higher application rates under Scenario E, compared to baseline, resulted in higher leached loads. Note that our modelled loads from the dunes included only that portion that flows towards Whangateau Harbour. A higher treated wastewater load flows towards the open coast.

Table 2: Unsaturated Zone Nitrogen Mass Balance Summary for Golf Course (upper) and eucalypt block (lower)

Month	Applied Load (kg-N)	Leached Load (kg-N)	Uptake Load (kg-N)	Denitrif. Load (kg-N)	Avg. Conc. (g-N m^{-3})	Min. Conc. (g-N m^{-3})	Max. Conc. (g-N m^{-3})
OBGC							
Jan	110	33	34	44	7	1	18
Feb	70	21	34	21	4	0	11
Mar	84	23	34	27	3	0	6
Apr	47	5	34	6	7	0	20
May	47	7	34	6	7	0	22
Jun	47	10	34	4	4	0	23
Jul	47	9	34	4	4	0	19
Aug	47	8	34	4	5	0	20
Sep	47	6	34	6	7	0	23
Oct	47	4	34	8	11	1	23
Nov	277	156	34	50	29	6	59
Dec	138	72	34	61	13	1	43
Annual Tot.	1,008	352 (35%)	414 (41%)	242 (24%)	9	0	59
Eucalypts							
Jan	21	0	2	35	0	0	1
Feb	6	0	2	12	0	0	0
Mar	14	0	1	11	0	0	0
Apr	31	0	1	26	0	0	0
May	32	0	2	23	0	0	1
Jun	136	5	2	48	1	0	3
Jul	202	24	2	120	2	2	3
Aug	103	19	2	138	2	2	3
Sep	122	8	2	110	2	1	2
Oct	127	5	2	128	2	1	2
Nov	83	0	2	103	1	1	2
Dec	27	0	2	60	1	0	1
Annual Tot.	908	63 (7%)	25 (3%)	818 (90%)	1	0	3

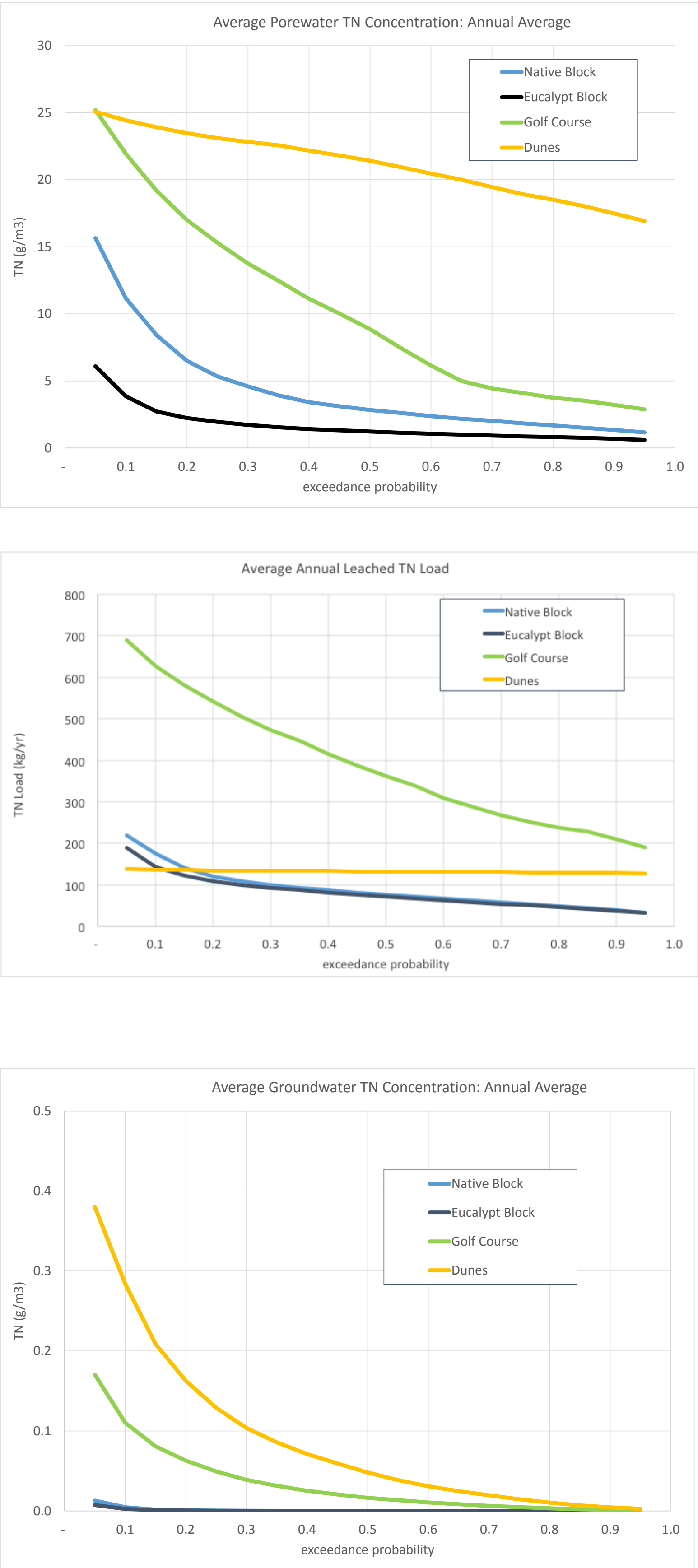


Figure 7: Modelled saturated zone N concentrations (top), N loads from unsaturated zone (middle) and saturated zone final N concentrations to Whangateau Harbour (bottom)

3.2.1 SENSITIVITY ANALYSIS

Our stochastic approach represents the best approach to deal with uncertainty, because rather than predicting a single value it predicts a range of outcomes from which the most probable (or the most extreme) can be selected. The passage to, and N removal from, saturated organic layers represents an increase in uncertainty, because we could not measure N removal rates in these deep organic layers. The groundwater modelling (Stuart et al., 2017) predicts passage through these layers with long residence times. Therefore, to quantify the influence of the residence time, we performed a sensitivity analysis on the predicted N concentrations leaving the subsurface saturated zone. We varied assumed residence time across an extended range of values to demonstrate model sensitivity to this uncertain parameter. For this exercise, the Eucalypt block model was used, in a deterministic mode, with all stochastic inputs set at median levels. Results (Figure 8) demonstrate that in biologically active subsurface zones, predicted groundwater N concentrations are highly sensitive to residence times below a threshold of approximately 200 days. For residence times greater than approximately 200 days, calculated concentrations are essentially negligible (< 0.05 g m^{-3}). In comparison, the estimated residence time from the groundwater modelling (Stuart et al., 2017) varied from 15-40 years.

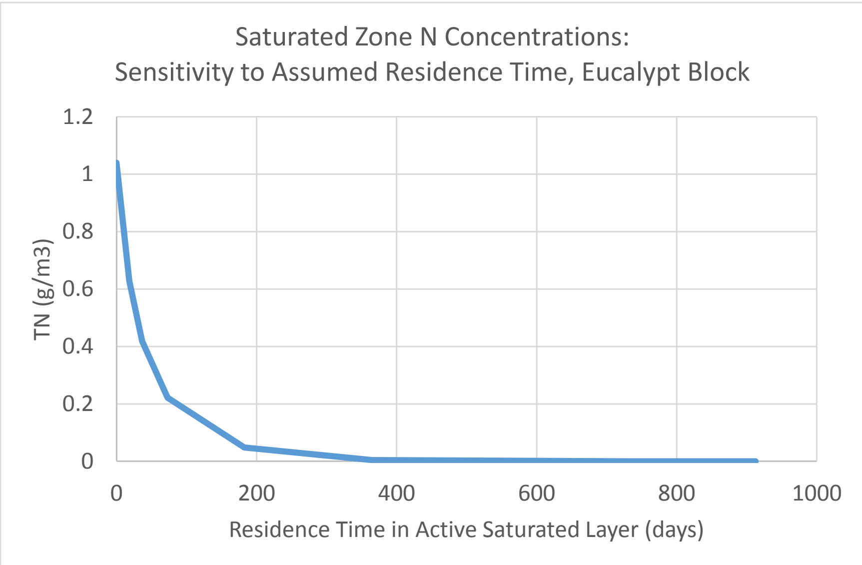


Figure 8: Sensitivity analysis of effects of residence time on predicted final N concentration

4 CONCLUSIONS

Our assessment has demonstrated: (i) that N losses via denitrification are significant, particularly within the deep peat layers prevalent on the Jones Rd site, (ii) based on measured biochemical rates and chemical measures our modelling showed that a maximum (90%ile) of ~1040 kg N/year would percolate from the unsaturated zone, with the most likely (median) figure being ~655 kg/year. This is equivalent to ~3.8% and 2.4% of the total annual N load to the harbour (Streamlined Environmental Ltd, 2014), respectively. The equivalent figures for the increased irrigation scenario are 1280 kg/y (90%ile) and 920 kg/y (median) equivalent to 4.6% and 3.3% of the total annual load respectively.

When further N transformations from the saturated organic layers identified by PDP (2015) is taken into account, the proportion of N contributed by WWTP irrigation reduces to ~ zero. This is the case both for the current irrigation load and the proposed increased irrigation load. We acknowledge, however, that there is more uncertainty associated with passage through these "active" saturated layers, but nevertheless, we estimate that overall the contribution of Omaha WWTP irrigation to Whangateau harbour N inputs is < 2.0%. It is likely that these layers are highly reduced which would favour dissimilatory nitrate reduction to ammonium (Tiedje, 1988) (see Figure 2). However, oxidation-reduction processes proceed in a fixed sequence and there will be a zone where redox conditions favour denitrifiers. Our sensitivity analysis suggests that in any case, only ~200 days is necessary to effect complete removal of leached NO_3-N. We note even if dissimilatory nitrate reduction to ammonium were the dominant process, the net result would be the same (i.e. N originating from the Omaha WWTP entering harbour waters would be minimal).

The study reported here was only one in a suite of studies commissioned by Watercare Services Ltd to support their consent application. The hydrodynamic study (MetOcean, 2016) demonstrated that the southern arm of the harbour was almost entirely flushed each tidal cycle. The combination of technical studies, excellent stakeholder consultation and support (Stuart et al. 2017) convinced the panel hearing the consent application to grant all consents for 35 years (with appropriate monitoring conditions)

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