Modelling the environmental effects of wastewater disposal at the Masterton land-based sewage effluent disposal scheme

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Report to Beca Carter Hollings & Ferner
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DISCLAIMER

This report has been prepared by The Horticulture and Food Research Institute of New Zealand Ltd (HortResearch), which has its Head Office at 120 Mt Albert Rd, Mt Albert, AUCKLAND. This report has been approved by:

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EXECUTIVE SUMMARY

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The Masterton District Council are currently reviewing their treatment and disposal options for effluent generated from Masterton's oxidation ponds. Through consultation, it has been decided that a mix-and-match option of irrigation to land and direct disposal to the river is the best way forward.

Beca Carter Hollings & Ferner Ltd has engaged the services of HortResearch to carry out a desktop modelling exercise to assess the potential impact of effluent disposal on leaching and runoff losses from the Homebush site. The purpose of this report is to provide an overview of the model calculations, describing the model's inputs and assumptions, and presenting a tabular summary of the model outputs.

Under the proposed scheme, municipal wastewater from the Masterton Township is piped directly to the site where it is first stored and treated in the oxidation ponds. The scheme will use a series of valves to control the flow of effluent from the pond. Border strip irrigation will be applied to pasture on the free-draining soils, while drip irrigation will be applied to short-rotation forest on the poorly drained soils. The pasture will be harvested using a cut-and-carry operation. The trees will be coppiced at four-year intervals. The purpose of regular harvesting is to remove excess nutrients (nitrogen (N) and phosphorus (P)) from the site. Preference will be for land application, although direct disposal to the river will be allowed in accordance with the adopted discharge rules. During rain events a wipe-off drain will collect all runoff water from each land area and a pump will return the first flush (i.e. 2 mm) to the storage pond. Thereafter, surplus runoff will flow directly into the Makoura Stream.

The original concept for the irrigation scheme, for which the modelling results are presented in this report, was for a border strip scheme on the well drained soils and a drip irrigation scheme on the heavier soils (17 hectares at the south west corner of the site). As part of the ongoing investigations for the proposed scheme, a drainage trial on the heavier soils was undertaken, with this work being concluded after the work on the water and nutrient balance had been completed. The results of the subsequent drainage trials showed that these heavier soils could potentially be irrigated at the higher rate of 10 mm/day in the summer and up to 5 mm/day in the winter with the use of a border strip scheme. In addition, the modelling was carried out using an oxidation pond leakage rate through the base of the ponds of 490 m^3/day. Subsequent to the completion of the water and nutrient balance (as detailed in this report), further leakage monitoring led to the adoption of revised leakage rates. For the ponds at their normal operating level, a leakage rate of 800 m^3/day with an upper bound of 1,700 m^3/day has been adopted. Leakage at higher ponds levels, simulating the effects of storage, provided an estimated leakage of 1,200 m^3/day with an upper bound of 2,400 m^3/day. The results of the modelling to determine the irrigation storage requirements, as detailed in this report, are therefore conservative for the following reasons:

- Higher irrigation rates are possible over the area of heavier soils compared with the application rates used in the model. This will have the effect of allowing a greater quantity of effluent to be applied as irrigation
- The higher rate of leakage will reduce the storage in the oxidation ponds.

Modelling has been carried out to determine both the storage requirements of the oxidation ponds and the environmental footprint from the land-based disposal. The calculations utilize a complete record of eight years of climate, river flow and influent data that are input to the SPASMO (Soil Plant Atmosphere System Model). Initially a base-case model was established. Then a range of different options was tested in order to assess the impact of altering various operational parameters. The outcome of the option testing was that Option 6 was selected as the preferred scheme. Key impacts for the preferred scheme are summarized below.

Discharge Rules

- Irrigation will occur whenever possible
- Discharge direct to river will occur:
  - In summer only when the river flow is greater than the median river flow
  - In winter only when the river flow is greater than the half median river flow
- Whenever there is a direct discharge to the river, the ratio of river flow to effluent flow will be at least 30 to one, up to the maximum effluent flow
- The maximum effluent flow for a direct discharge is 104,000 m^3 d^{-1} (1200 L s^{-1}).

Storage Volume

- The maximum operating storage volume is 339,000 m^3, which is 24,000 m^3 less than the maximum storage capacity of the current oxidation ponds (i.e. 363,000 m^3)
- This margin (of 24,000 m^3) is equivalent to a water depth of approximately 0.1 m, or about two days of inflow to the ponds, and is considered to provide a prudent degree of buffer capacity.

Disposal of Effluent under the Preferred Scheme

- Over the eight years of record, disposal by irrigation occurs on 92% of days with approximately 21% of the inflow volume being disposed of by irrigation
- Over the eight years of record, a direct discharge to river occurs on approximately 63% of days, with approximately 77% of the inflow volume being discharged directly to the river.

Impact of Effluent used as Irrigation

The fate of nitrogen, phosphorus and bacteria (*Escherichia coli*) in the upper soil layers was assessed as part of the model development. An estimate was made of the quantity and quality of the water that would leach to the underlying groundwater, with further work on the assessment of the impact on groundwater quality being undertaken and reported separately by Pattle Delamore Partners.

Key conclusions with respect to these parameters are summarised below.

Nitrogen

Modelling shows leaching of nitrate nitrogen to be of little concern.

- Each year, on average, the effluent irrigation adds about 231 kg N ha^{-1} to the pasture zones and 90 kg N ha^{-1} to the short-rotation forest
- The annual nitrogen uptake by pasture is calculated to be between 300-415 kg N ha^{-1} and this includes an annual nitrogen fixation of between 156-180 kg N ha^{-1} based on a

15% clover content. The corresponding value of nitrogen uptake by the short-rotation forest is calculated to be about 90 kg N ha^{-1}

- The cut-and-carry operation will remove most of the pasture dry matter (DM; between 75-85%) and nutrients from the site
- The average nitrate-nitrogen concentration of drainage water at 1.0 m is calculated to be 2.7 mg L^{-1}. This is about four times lower than the current New Zealand Drinking Water Standard (MFE 2005) and therefore should not pose a significant threat to the quality of the groundwater.

Phosphorus

Phosphorus accumulation and leaching has been assessed for a 28-year period (based on available climate and river data), with the results showing:

- Each year, on average, the effluent irrigation adds about 50 kg P ha^{-1} to the pasture zones and 20 kg P ha^{-1} to the short-rotation forest
- The annual uptake of P by pasture is calculated to be between 20-35 kg N ha^{-1} for the pasture zones and about 5 kg P ha^{-1} for the short-rotation forest. Thus, the pasture takes up about half the applied phosphorus, which is largely removed from the site, while the trees take up about 25%
- It is calculated that the maximum soil P concentration will remain <25% of the phosphorus sorption capacity. Although the solution concentration entering the ground water could slowly rise to 0.2 mg L^{-1}, additional dilution in the ground water, combined with strong adsorption by the deep clay-rich layers, means the off-site impacts on surrounding ground water are likely to be negligible.

Bacteria (E. coli)

Bacteria transport through the soil has been assessed for a 28-year period, with the results showing:

- Between 95-99% of surface-applied *E. coli* are removed during transport through the top 1 m of soil. Inactivation (die off) will account almost all the applied bacteria
- None of the simulations predicts a significant accumulation of bacteria numbers over time. The average concentration in the drainage water at a depth of 1 m sometimes exceeds New Zealand Drinking Water Standards of 1 cfu per 100 ml by a factor of between 1 and 15. However, additional die off and dilution in the groundwater is expected to reduce these concentrations further
- It is concluded that *E. coli* in treated effluent added to land is unlikely to have a detrimental impact on the quality of the groundwater under the disposal site.

The modelling has shown that the land area provided for disposal is able to accommodate the water and nutrient loads being applied and provide an effective filtering capacity. However, three conditions will need to be taken into account:

- The hydraulic capacity of the soils should not degrade under irrigation
- The river flow characteristics should remain similar to past years (i.e. there should be no severe and prolonged droughts that alter the duration of low flows in summer)
- If effluent volumes and composition change compared with past years, then the model outputs will also change.

Predicting how these characteristics might change over time is beyond the scope of this desk-top modelling exercise, although this could be considered in the future.

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1. BACKGROUND

Effluent from the Masterton wastewater treatment plant (WWTP) is currently disposed of to the Ruamahanga River, via the Makoura Stream. The Resource Consent for this activity is due to expire in 2010, by which time a new consent is required to be in place and a long-term upgrade complete.

Beca Carter Hollings & Ferner Ltd (Beca) has been engaged by MDC to carry out the design of the upgrade, which includes obtaining the new Resource Consent. The upgrade will involve constructing maturation cells within the existing secondary oxidation pond to achieve an improved effluent quality and the addition of an irrigation scheme. Effluent will then be disposed of by a combination of irrigation and/or direct discharge to the Ruamahanga River.

In order to support the resource consent application, an Assessment of Environmental Effects (AEE) is being prepared, and in parallel, the preliminary design of the scheme is being undertaken.

Several sub-consultants have been engaged by Beca to provide expertise to support the preparation of the AEE and preliminary design. HortResearch has been engaged to provide specialist expertise with respect to modelling of the water balance for the scheme and an assessment of the impact of nutrient and bacterial loading on the irrigation area.

2. SCOPE OF THIS STUDY

HortResearch has been engaged to carry out a desktop modelling exercise to assess the key outcomes arising from the disposal of effluent by a combination of land treatment (irrigation) and direct discharge to the Ruamahanga River. These key outcomes are:

- To calculate the impact of effluent disposal on the leaching and runoff losses of water, nutrients (i.e. nitrogen (N) and phosphorus (P)) and contaminants (i.e. *Escherichia. coli* bacteria) from land at the Homebush site
- To determine the volume of storage that accumulates in the oxidation ponds in response to a set of discharge rules that dictate when a direct discharge to the Ruamahanga River is permitted, and also taking into account that the priority method of disposal for effluent is by irrigation whenever soil conditions allow this to occur during summer and winter.

For this task, the computer model SPASMO (Soil Plant Atmosphere System Model) was adapted to calculate a water and nutrient balance for the site, and to estimate the storage requirement for the ponds. A full description of the SPASMO model is presented in a number of scientific papers (e.g. Green et al., 1999; Rosen et al. 2003; Sharma et al., 2005) and client reports to regional councils (e.g. Green et al 2002) and industry (e.g. Green et al 2003b).

A schematic diagram of the planned disposal scheme is provided in Figure 1. Municipal wastewater from the Masterton Township is piped directly to the site where it is treated in the oxidation ponds. Currently there are three ponds at Homebush, although the model treats these as a single volume with a known surface area (see Appendix A for a calculation of the effective surface area and the depth of the ponds). A water balance for the storage pond is calculated by considering the volumes of water going into (i.e. as influent, rainfall and return flow) and out of the ponds (i.e. as evaporation, leakage losses through the base of the ponds, irrigation to land and direct discharge to the river).

The proposed scheme will use a series of valves to control the flow of effluent, and these will be operated independently. For the purpose of modelling, the total land area has been divided into 11 zones, each zone representing an area of like soils. Border strip irrigation was applied to pasture on 10 of the free-draining zones (labelled as L1 to L10 in Figure 1). The remaining zone (L11) is on the poorer soils planted in a short-rotation forest and effluent volumes were applied via a dripper irrigation line. The pasture was harvested using a cut-and-carry operation. The tree site was coppiced at four-year intervals. The purpose of regular harvesting was to remove nutrients (N and P) from the site. Preference was for land application, although direct disposal to the river was allowed in accordance with the adopted discharge rules, which are detailed in Section 4. A wipe-off drain collects all runoff water from each land area and a pump will return the first flush (i.e. 2 mm) of runoff to the storage pond. Thereafter, surplus runoff flows directly into the Makoura Stream.

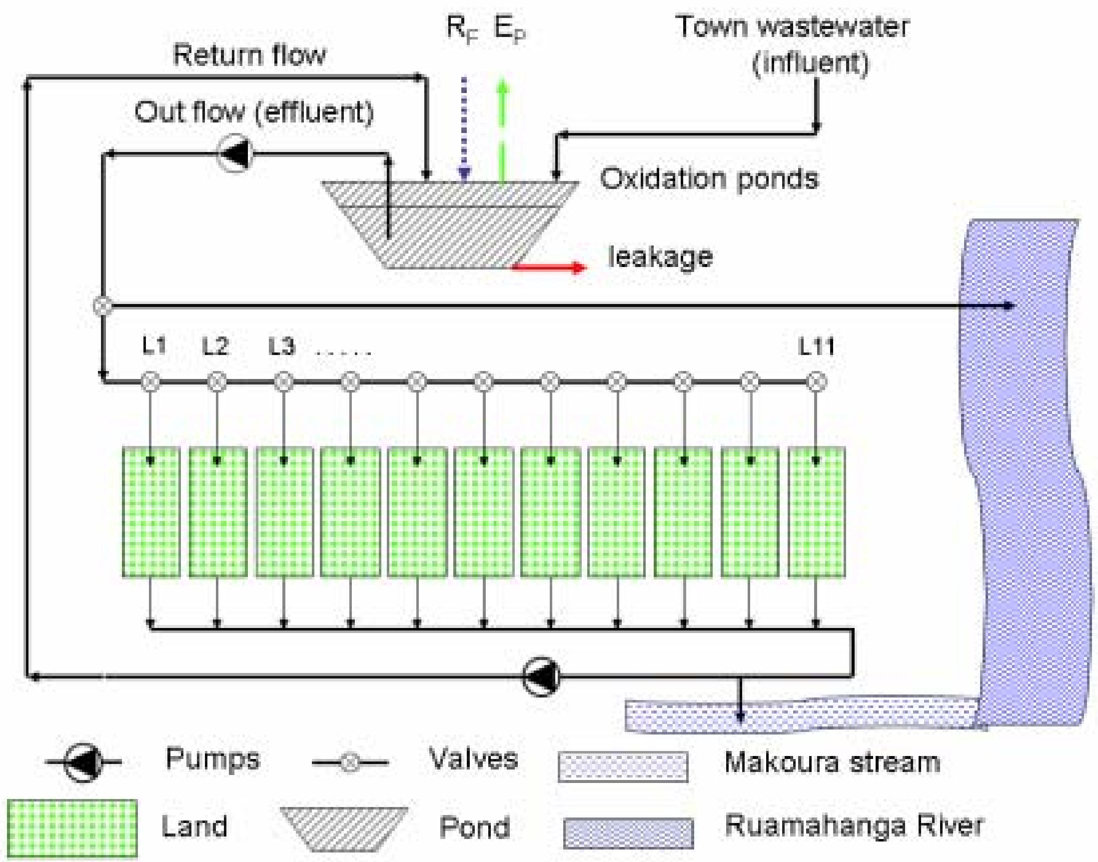


Figure 1. Schematic of the proposed irrigation scheme at the Homebush site. For the development of the model, irrigation is controlled by a series of valves that can be operated independently. A wipe-off drain collects runoff water from each land area and returns it to the storage pond, or lets it flow into the Makoura stream. Here R_F = rainfall and E_P = evaporation.

A tailored computer model was developed to calculate the site's water and nutrient balance, including evaporation losses and storage volumes of the pond as well as runoff and drainage losses from the site. Subroutines from our SPASMO model, which normally calculates the water and nutrient budgets for a single 'paddock', were linked to additional computer code that operates across a number of pre-defined land areas. The calculations run on a daily time step. Inputs include a time series of climate, river flows and effluent volumes. The site is divided into 11 zones of similar soil properties. Under the current design rules for operation, different amounts of effluent will be disposed onto each land area depending on the soils' hydraulic capacity to store and drain the water. The water balance of the oxidation pond is guided by a set of decision rules for disposal options. The operational rules for land and river disposal are described in Section 4.

The model calculations were run on a daily time step and the following predictions were generated:

- Volumes discharged to land. Land disposal occurs only when soil moisture and irrigation management permits it
- Volumes discharged to the Ruamahanga River. Effluent from the ponds is discharged directly into the river only when river flows are greater than a given trigger flow
- Storage volumes in the oxidation ponds. Effluent is accumulated in the ponds whenever irrigation and discharge to the river is less than the inflow volume
- The fate of nitrogen, including growth and N-uptake by pasture and the short-rotation forestry, and the quantity of nitrogen resident in the soil and leached to groundwater
- The fate of phosphorus, including P-uptake and the quantity of phosphorus resident in the soil and leached to groundwater
- The fate of bacteria (i.e. *E. coli*), including the filter capacity of the soil and the quantity of bacteria (cfu per 100 ml) leached to groundwater.

The goal of the modelling was to determine the outputs detailed above for a range of option simulations that would help to guide the decision regarding the preferred disposal scheme. To achieve this task, a range of options was set up by altering irrigation rates and rules for disposal. Other factors being considered by the modelling included the maximum storage volume under a given set of disposal rules, and the potential of ground water contamination from effluent containing nutrients (i.e. N and P) and bacteria leaching through the soil profile. Model output included a time series of nutrient and bacterial concentrations at a soil depth of 1.0 m that coincides approximately with the groundwater depth. Time series data were subsequently passed onto Pattle Delamore Partners (PDP) Ltd for their assessments of groundwater effects. The purpose of this report is to describe the modelling work undertaken, to explain the key assumptions and to provide supporting information where necessary.

The original concept for the irrigation scheme, for which the modelling results are presented in this report, was for a border strip scheme on the well drained soils and a drip irrigation scheme on the heavier soils (17 hectares at the south west corner of the site). As part of the ongoing investigations for the proposed scheme, a drainage trial on the heavier soils was undertaken, with this work being concluded after the work on the water and nutrient balance had been completed. The results of the drainage trial showed that these heavier soils could potentially be irrigated at the higher rate of 10 mm/day in the summer and up to 5 mm/day in the winter with the use of a border strip scheme. In addition, the modelling was carried out using an oxidation pond leakage rate through the base of the ponds of 490 m^3/day. Subsequent to the completion of the water and nutrient balance (as detailed in this report), further leakage monitoring led to the adoption of revised leakage rates. For the ponds at their normal operating level, a leakage rate of 800 m^3/day with an upper bound of 1,700 m^3/day has been adopted. Leakage at higher ponds levels, simulating the effects of storage, provided an estimated leakage of 1,200 m^3/day with an upper bound of 2,400 m^3/day. The results of the modelling to determine the irrigation storage requirements, as detailed in this report, are therefore conservative for the following reasons:

- Higher irrigation rates are possible over the area of heavier soils compared with the application rates used in the model. This will have the effect of allowing a greater quantity of effluent to be applied as irrigation
- The higher rate of leakage will reduce the storage in the oxidation ponds.

3. SITE CHARACTERISTICS

The first task for the modelling was to determine appropriate soil properties (i.e. soil texture, drainage class, water retention, depth to gravel, etc.) at the site. This information was gleaned from a number of studies carried out by Landcare Research (Wilde & Dando 2004, 2005). The following is a summary of their findings.

The Homebush site occupies approximately 90 ha on low terraces and a former floodplain of the Ruamahanga River immediately west of the river (Figure 2). The land is currently in dry-land pasture except for a small patch (~2 ha) of native bush on the southwest corner. The soils have been formed from the river alluvium comprising gravelly sediments overlain by predominantly sandy and silty alluvial sediments. Nearer the river, the alluvium is coarser, with a tendency to have sandy textures sometimes interspersed with gravels plus fine sandy loams, sandy loams and gravelly sand textures as well as some loamy silts. These soils are named Greytown sandy loam and gravelly sandy loam in the Soil Survey of the Wairarapa Valley (Heine 1975). Westward of these coarser soils, a finer-textured sediment overlies the gravels, and the soils are silty-textured (silt loam and silty clay loam textures) with intermittent clay-rich layers at depth. These soils are named Greytown silt loam (Heine 1975).

More than 30 soil pits and auger holes have been excavated across the Homebush site to identify soil textural properties and determine the soil depth to the gravels (see Appendix B for locations of the sampling sites). A general soil map of the site was then developed by grouping ‘like soils’ with similar textures, infiltration rates and water storage capacities. The proposed irrigation design utilizes this detailed soil information to divide the whole land area into 11 zones that can be isolated and operated independently of the others (Figure 2). Each zone represents an area where the soil’s hydraulic and physical properties are similar. The northern land area is generally free draining, while the southwest area is generally poorly draining. Border-strip irrigation will be applied to pasture on the free-draining soils, and drip irrigation will be applied to short-rotation forestry (*Eucalyptus ovata* trees) on the poorly-drained soils.

In addition to soil textural and drainage class information, Landcare Research also carried out a detailed analysis of the soils’ hydraulic and physical transport properties. Their measurements included infiltration rates (how fast the water moves though the soil), water retention capacities (how much water the soil holds), and a mineral and chemical analysis for N and P (Wilde & Dando, 2004, 2005). Table 1 summarizes basic soil textural information for each of the irrigation zones, while Table 2 presents data describing the soils’ capacities to store and transport nutrients. Where clay-rich soil materials occur at shallow depth, interpedal cracks and macropores are able to conduct water at significant rates (>77 mm h^{-1}), and so initial surface ponding should not be an issue. Where there are deeper clay-rich layers, the soils will tend to conduct water at much slower rates, of between 0.5-4.0 mm h^{-1}, mainly because there are fewer macropores at depth. Those soils with more silty clay material at depth (e.g. site 11) are classified as poorly drained, and they will accommodate a much lower hydraulic loading.

Table 1. Profile of 'average' soil texture, as determined from a visual assessment of soils across the 11 sites at the Homebush, Masterton site (data from H. Wilde, Landcare Research).

Depth [cm]	site										
	1	2	3	4	5	6	7	8	9	10	11
0-10	loamy silt	silt loam	silty clay loam	loamy silt	silt loam	loamy silt	loamy silt	silt loam	silt loam	silt loam	clay loam
10-20	loamy silt	silty clay loam	silt loam	loamy silt	silt loam	loamy silt	loamy silt	silt loam	silt loam	silt loam	clay loam
20-30	loamy silt	silty clay loam	silt loam	loamy silt	sandy loam	loamy silt	loamy silt	silt loam	silt loam	silt loam	silty clay
30-40	silt loam	silty clay loam	silt loam	silt loam	sandy loam	loamy silt	loamy silt	sand	sand	sand	silty clay
40-50	silty clay loam	silty clay loam	silty clay loam	silt loam	sandy loam	loamy silt	loamy silt	loamy silt	sand	sand	silty clay
50-60	silty clay loam	silt loam	silty clay loam	silt loam	silt loam	loamy silt	loamy silt	loamy silt	sand	sand	silty clay
60-70	silty clay	sandy loam	silty clay	loamy silt	loamy silt	loamy silt	silt loam	silt loam	sand	sand	silty clay
70-80	clay loam	sandy loam	silty clay	loamy silt	loamy silt	sandy loam	silt loam	silt loam	silty sand	silty sand	silty clay
80-90	sandy clay loam	sand	silt loam	loamy silt	silty clay loam	sandy loam	sandy loam	silt loam	sand	sand	silty clay
90-100	sandy loam	sand	loamy silt	clay loam	silty clay loam	silty clay loam	sandy loam	silt loam	sand	sand	silty clay

Soil pH ranges from moderately acid on the wetter soils in the south of the property to near neutral in the north. Mineralizable nitrogen, which results from activities of the soil's microbial biomass, varies quite markedly across the property, and is generally quite low. Total organic carbon is also very low on all soils, although the carbon to nitrogen (C/N) ratio is typical of soils under pasture. Olsen P values (0-10 cm) vary across the site, being generally adequate to low. Pasture and tree growth is expected to be water and nitrogen limited on these soils under natural rain-fed conditions.

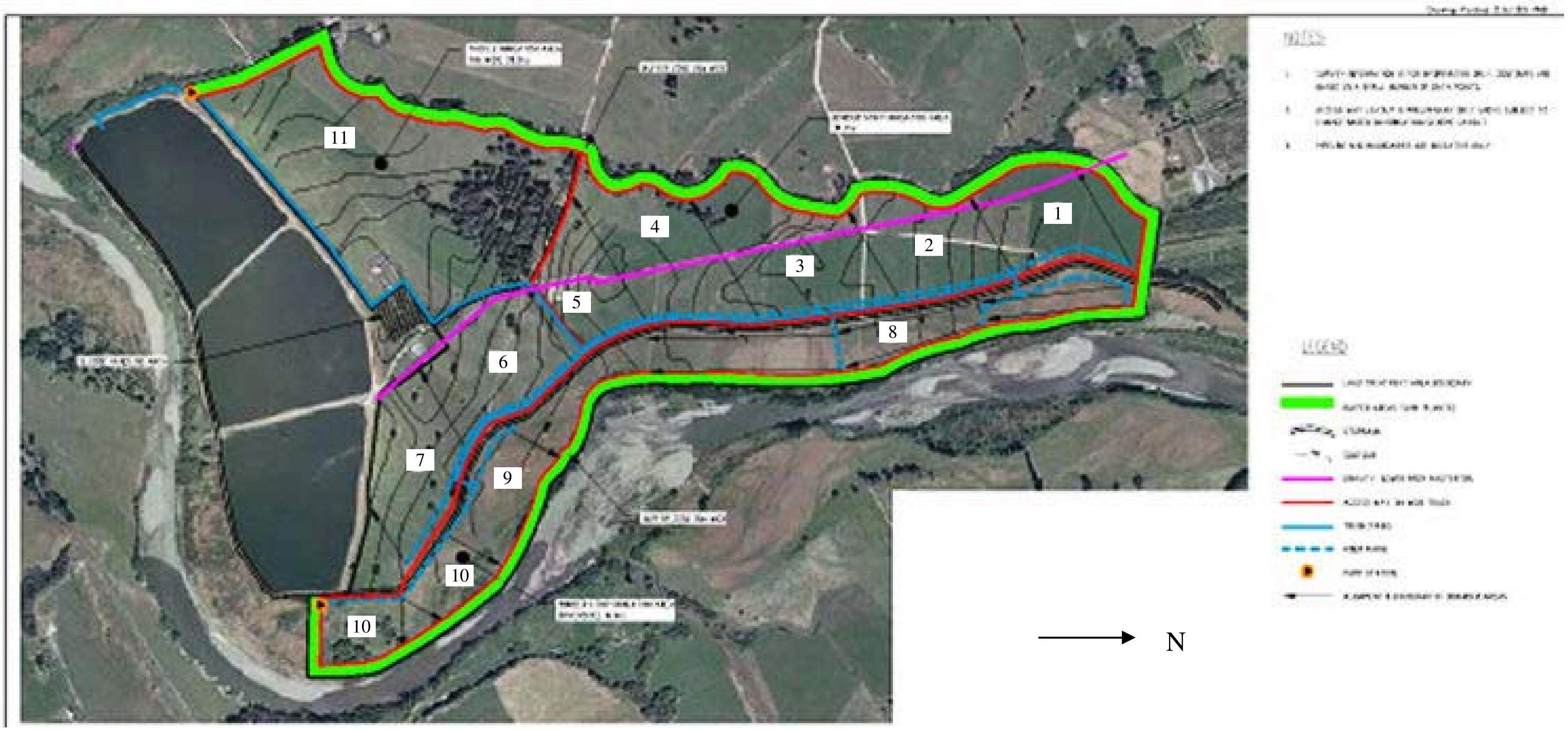


Figure 2. Plan of the proposed irrigation layout at the Homebush site (supplied by Beca). The disposal area is divided up into a total of 11 irrigation zones that can be isolated and operated independently of the others. Zones 1 to 10 are cut-and-carry pasture on free draining soils. Zone 11 is short-rotation forest on poorly drained soils.

Table 2. Model parameters used to relate soil texture to the soil's hydraulic properties. Data were sourced from Landcare Research (LCR) reports (Wilde & Dando 2004, 2005) and the New Zealand Soils Database (NZSDB, Landcare Research). Here SAT, FC and WP refer to the soil's water content at saturation, field capacity and wilting point; RAW and TAW are the corresponding values of readily available and total-available soil water; and K_{SAT} is the saturated hydraulic conductivity.

Texture	Data source	Bulk density [kg L^{-1}]	SAT [L L^{-1}]	FC [L L^{-1}]	WP [L L^{-1}]	RAW [L L^{-1}]	TAW [L L^{-1}]	K_{SAT} [mm d^{-1}]
clay / silty clay	from NZSDB	1.03	61.6	59.3	48.3	3.7	11.0	14
silty clay loam	Average (LCR)	1.39	47.9	35.6	16.6	9.5	19.0	30
clay loam	Bush (LCR)	1.16	57.0	35.3	15.2	11.3	20.1	43
loamy silt	Northeast Bore (LCR)	1.36	49.4	28.9	8.7	11.5	20.1	81
silt loam	Average (LCR)	1.48	44.2	37.4	20.0	7.8	17.4	86
sandy clay loam	from NZSDB	1.27	53.3	39.7	19.5	10.1	20.2	96
sandy loam	Enclosure (LCR)	1.37	49.0	26.1	9.2	9.6	16.9	540
loamy sand	Pumphouse (LCR)	1.38	48.8	24.1	5.1	16.7	19.0	1410
sand	from NZSDB	1.42	47.4	15.3	1.1	10.5	14.2	2870
gravel	from NZSDB	1.53	43.7	28.2	17.4	5.5	10.8	5400

4. MODELLING APPROACH AND ASSUMPTIONS

The water and nutrient balance of the site was calculated using an extended version of HortResearch's SPASMO model (Green et al. 2003b). This model uses appropriate science to link the mechanisms of water and nutrient flow through soil with the complex transformations that result from both natural processes that occur in soils and plants, as well as those processes consequent upon the surface application of effluent to soil. The SPASMO model is described in more detail in Appendix A. Field validations of SPASMO include nitrate leaching under pasture (Green et al. 2005; Rosen et al. 2003) and fruit crops (Green et al. 2006), and pesticide movement (Sharma et al. 2005; Sharma et al. 2006) under a range of New Zealand soils and climatic conditions.

For this project, additional mathematical routines were developed within SPASMO to enable a prediction of phosphorus and microbial transport through the soil. Furthermore, an additional 'master' programme was developed to connect subroutines in SPASMO with new routines to accommodate the simultaneous disposal of effluent across multiple land areas, and a range of options for effluent disposal onto the land and into the river. All simulations were based on daily climate records from Te Ore Ore (1977-2005). Soil properties were derived using observations from auger hole sampling and soil pits excavated from across the site (Tables 1 and 2).

The water balance within each irrigation zone was calculated by considering inputs (rainfall and irrigation) and losses (plant uptake, evaporation, runoff and drainage) of water from the soil profile. Plant growth and nutrient (N and P) uptake was calculated from daily values of solar radiation and mean air temperature, with growth being reduced whenever soil water or nutrients were limiting. The water-borne transport of bacteria through the soil profile was modelled as a colloidal-filtration process that depends on soil texture and an inactivation (die-off) rate that is determined by a characteristic half-life (residence time). These abiotic processes were modified by the temperature and water content of the soil. Details of the modelling approach are described in more detail in Appendix A.

DEVELOPMENT OF THE MODEL

The model has been developed using eight years of actual records of relevant input parameters (the input parameters are described in Appendix B). The modelling work has involved testing a range of options based on a set of discharge rules (for irrigation and direct discharge to the river), with the key output for each option being the derivation of a maximum operating storage volume. The maximum operating storage volume is required to be less than maximum storage capacity of the oxidation ponds, which has been set at 363,000 m^3 (i.e. 363,000 m^3 more than the minimum working volume) based on the need to achieve a satisfactory factor of safety with regard to embankment stability. The maximum storage capacity has been defined by Beca.

A key focus of the development of the discharge rules was to recognise the importance of the Ruamahanga River for its recreational value, particularly during summer at times of low river flow when there is a strong community desire to use the river for contact recreation. Specific aspects relating to the model are discussed below.

The Ponds

For the purpose of modelling, the whole pond system was treated as one volume with an effective surface area of 26 ha. The minimum pond working depth was set equal to $Z_P = 1.8$ m and the initial volume was set equal to $V_P = 461{,}000$ m^3. The pond had a sloping side with a ratio of $\Delta = 1{:}\ 2.5$, and the base width was set equal to W ~502 m. The pond volume at any depth Z was then calculated as $V_Z = [Z_P[W^2 + (W{+}2\Delta Z_P)^2\ ]^{0.5}]/2$. This equation enables an estimate of pond depth for any given storage volume. Pond storage, S m^3, was calculated from $S = V_Z\text{-}V_P$. Evaporation loss from the pond was calculated using Equation A8 (Appendix A) based on historical records of daily climate taken from the NIWA climate station at Te Ore Ore, near Masterton. An allowance of 490 m^3 of effluent was assumed to leak from the base of the oxidation pond. This estimate is based on data reported previously by Beca (Beca 2005). This equates to drainage rate of about 2 mm d^{-1}. Any runoff that is generated from individual irrigation areas will be captured by a wipe-off drain and some of this will be pumped directly back into the pond.

Discharge Rules: Direct Discharge to River

The following rules relating to a direct discharge to the river were utilised in the model option testing:

- A direct discharge was permitted only when the Ruamahanga River reached a 'trigger flow'. The trigger flows used were:
  - Median river flow of 12.3 m^3 s^{-1}
  - Half median river flow of 6.15 m^3 s^{-1}.

The median river flow was calculated from an analysis of daily river flows at Wardell's Bridge between the years 1989-2004 (data supplied by Beca). It is also noted that some of the options had a zero trigger flow:

- A dilution factor of at least 30 times was maintained (i.e. the ratio of Ruamahanga River flow: effluent discharge rate >30:1) for all effluent discharge rates up to the maximum value

- The maximum effluent discharge rate was set at 104,000 m^3 d^{-1} (1200 L s^{-1})
- In some options, a minimum volume of effluent was discharged.

Discharge Rules: Disposal by Irrigation (Land Treatment)

The following rules relating to irrigation of effluent on to land were utilised in the model option testing:

- Preference was given to effluent irrigation. If a site could be irrigated, then the maximum volume possible, under the rules, was applied to the land
- Irrigation to the border strips (Zones 1 to 10) was set at 100 mm d^{-1} in the summer and 50 mm d^{-1} in the winter with a minimum 10-day stand down period between irrigations. The irrigation rates adopted as the basis for the design were based on the recommendations of a number of experts in the fields of soil science and irrigation (including specific expertise with irrigation of effluent from wastewater treatment plants) who were engaged to provide input into the project. In some options, higher and lower application rates were also considered
- Irrigation to the short-rotation forest (Zone 11) was via drippers set at 5 mm d^{-1} in the summer (every day) and zero in the winter. In some options, higher and lower application rates were also considered
- The maximum amount of irrigation applied daily to the land was not allowed to exceed the soil capacity on that day. On those days when rainfall occurred, irrigation was set equal to the lesser of the target rate or the soil's total storage capacity (calculated as drainage plus refill depth) minus the amount of rainfall on that day
- On those days when the rainfall exceeded 20 mm, all pastures were given an additional two-day stand down period before the next irrigation resumed
- Storm water was separated from effluent runoff. When a rainfall event occurred, any runoff from the land area being irrigated on a given day was always directed back to the storage pond. The first 2 mm of rainfall from the other paddocks not being irrigated was also directed back to the ponds. Thereafter, any excess runoff was otherwise directed into the Makoura stream.

Option Analysis

Calculations were carried out for 16 options in order to examine the effects of different irrigation rates and return periods on the storage requirements for the oxidation pond and the fate of nutrients and contaminants contained in effluent applied to the land. For the purpose of calculation, the year was divided into two periods. Summer was classified as the period between 1 November and 30 April inclusive. The remaining 'winter' period ran from 1 May to 31 October. Disposal rules were different for the summer and winter periods. For the purpose of assessing and reporting various options, a 'base case' option was used as the reference point. Other options were then described by changes made to the base case.

The 'base case' option for irrigation (referred to as Option 0) was modelled in the following way:

- Summer trigger flow of median river flow, 12.3 m^3 s^{-1}
- Winter trigger flow of zero
- Summer minimum discharge volume of 35,000 m^3 d^{-1}
- Winter minimum discharge volume of zero
- Maximum effluent discharge of 100,000 m^3 d^{-1}
- Leakage through the base of the ponds of 490 m^3 d^{-1}
- Winter drip irrigation rate of zero
- Winter border strip irrigation rate of 50 mm with minimum 10 d return

- Summer drip irrigation rate of 5 mm d^{-1}
- Summer border strip irrigation of 100 mm with minimum 10d return.

Additional options were established by altering some of the base-case parameters as detailed below (Table 3 presents a summary of the options considered).

Option 1: Same as the base case, but with fewer discharges to the river in the wintertime. Under this options the winter trigger is set at a minimum flow of 12.3 m s^{-1} (and the minimum disposal volume is kept at 35,000 m^3 d^{-1})

Option 2: Same as the base case, but with larger irrigation volumes applied to the more freely-draining areas. In this case, 150 mm of irrigation was applied every 10 days in summer and 75 mm was applied every 10 days in winter for pasture sites on Zones 7 – 10 only

Option 3: Same as the base case, except that larger volumes of irrigation are applied to pasture on Zones 7-10, and the minimum return period for irrigation was set at just 5 days. Under this option 75 mm was applied in summer and 37.5 mm was applied in winter

Option 4: Same as the base case, but with larger irrigation volumes applied to all the pasture sites (i.e. Zones 1-10). Under this option, some 150 mm is applied in the summer, and 75 mm is applied in the winter to Zones 1-10, and the minimum return period for irrigation is set at 10 days

Option 5: Same as the base case, but with shorter return period for irrigation volumes applied to all pasture sites. Under this option, 50 mm is applied in the summer, 25 mm is applied in the winter, and the minimum return period for irrigation to the border strip areas is set at 5 days.

Option 6: Same as the base case, but with the winter trigger is set at a minimum flow of 6.15 m s^{-1} and the maximum discharge volume set at 104,000 m^3 d^{-1}. This option was eventually chosen as the preferred option (see Section 5)

Option 7: Same as base case, but larger irrigation volumes are applied to all the pasture sites (i.e. Zones 1-10). Under this option, some 120 mm is applied every 10 days in the summer and 60 mm every 10 days in the winter to the border strip areas

Option 8: Same as base case, but smaller irrigation volumes are applied to all the pasture sites (i.e. Zones 1-10). Under this option, some 70 mm is applied every 10 days in the summer and 35 mm every 10 days in the winter for the border strip areas

Option 9: Same as the base case, but with each pasture site rested for at least 2 weeks before harvesting. This option is designed to test the impact of having to rest the soil before the cut-and-carry operation can proceed

Option 10: Same as the base case, but using hourly river data to determine if river discharge can occur, as opposed to using the daily average river flow as the basis for determining when a trigger flow is reached. If the river flow each hour is greater than 12.3 m^3 s^{-1}, then the allowable discharge to the river equals the sum of the hourly flow divided by the dilution factor (=30). Otherwise no discharge is allowed. This rule is limited by the maximum river discharge volume of 104,000 m^3 d^{-1}

It is noted that in the practical operation of the scheme, the intention is that when a trigger flow is reached, a direct discharge to the river will occur.

Option 11: Same as the base case, but limiting the river discharge in summer to those times when the mean daily river flow >14 m^3 s^{-1}. This option is designed to test the impact of a different trigger flow

Option 12: Same as the base case, but limiting the river discharge in summer to those times when the mean daily river flow >16 m^3 s^{-1}. This option is designed to test the impact of a different trigger flow

Option 13: Same as the base case, but limiting the river discharge in the summer to those times when the mean daily river flow >18 m^3 s^{-1}. This option is designed to test the impact of a different trigger flow

Option 14: Same as the base case, but assuming leakage from the base of the oxidation pond is 50% higher. Under this option, leakage is 735 m^3 d^{-1}, which is equivalent to an average rate of 3 mm d^{-1} across the base of the ponds

Option 15: Same as the base case, but assuming leakage from the base of the oxidation pond is 50% lower. Under this option, leakage is 245 m^3 d^{-1} , which is equivalent to an average rate of 0.5 mm d^{-1} across the base of the ponds

Option 16: Same as the base case, but applying twice as much water to the poorly drained soils planted in short-rotation *Eucalyptus* trees. Under this option, the trees are irrigated using 10 mm d^{-1} (summer only).

Table 3. Parameter settings for the range of model options used to determine the preferred wastewater disposal scheme (Option 6) at the Homebush, Masterton site.

Option Number	Summer trigger flow [m^3 s^{-1}]	Winter trigger flow [m^3 s^{-1}]	Min. discharge (summer) [m^3 d^{-1}]	Min. discharge (winter) [m^3 d^{-1}]	Maximum effluent volume [m^3 d^{-1}]	Pond leakage rate [m^3 d^{-1}]	Drip irrigation (winter) [mm d^{-1}]	Border irrigation (winter) [depth (mm)/return period (days)]	Drip irrigation (summer) [depth (mm)/return period (days)]	Border irrigation (summer) [depth (mm)/return period (days)]	Other conditions
0	12.3	0	35,000	0	100,000	490	0	50/10	5/1	100/10	-
1	-	12.3	-	-	-	-	-	-	-	-	-
2	-	-	-	-	-	-	-	75/10[A]	-	150/10[A]	-
3	-	-	-	-	-	-	-	37.5/5[A]	-	75/5[A]	-
4	-	-	-	-	-	-	-	75/10	-	150/10	-
5	-	-	-	-	-	-	-	25/5	-	50/5	-
6	-	6.15	-	-	104,000	-	-	-	-	-	-
7	-	-	-	-	-	-	-	60/10	-	120/10	-
8	-	-	-	-	-	-	-	35/10	-	70/10	-
9	-	-	-	-	-	-	-	-	-	-	B
10	-	-	-	-	-	-	-	-	-	-	C
11	14.0	-	-	-	-	-	-	-	-	-	-
12	16.0	-	-	-	-	-	-	-	-	-	-
13	18.0	-	-	-	-	-	-	-	-	-	-
14	-	-	-	-	-	735	-	-	-	-	-
15	-	-	-	-	-	245	-	-	-	-	-
16	-	-	-	-	-	-	-	-	10/1	-	-

A irrigation rates were altered on Zones 7-10 only
B each pasture site rested for at least two weeks before harvesting
C used hourly river data to determine if river discharge can occur
\- indicates same value as Option 0.

5. MODEL OUTPUT AND DISCUSSION

Modelling was carried out to determine an appropriate operational scheme that would be both practicable and environmentally sustainable. Here ‘practicable’ means a reasonable storage volume and ‘environmentally sustainable’ means the risk of groundwater contamination is small. SPASMO was run for the 16 irrigated options to simulate the effect of different hydraulic and nutrient loads to the land. Factors being considered included the storage volumes of the oxidation ponds, under a given set of disposal rules, and the risk of ground water contamination due to excess water containing nutrients (i.e. N and P) and bacteria being transported through the soil profile.

Model output included daily volumes of effluent stored in the oxidation pond, the volume of effluent applied as irrigation, the amounts of water and contaminants (N, P and bacteria) leached to the groundwater, and the volumes of treated effluent discharged into the Ruamahanga River. For comparison of environmental effects, a non-irrigated option was also established (cut-and-carry pasture) to represent a background level for water and nutrient fluxes from a dry-land farm.

SELECTION OF THE PREFERRED OPTION

As previously discussed, the key requirement in the selection of a preferred scheme is that the maximum operating volume is less than the assessed maximum storage capacity of the ponds of 363,000 m^3. It is also necessary for the relevant discharge rules to reflect a strong environmental enhancement, particularly with respect to recreational use of the river in summer.

A summary of the volumetric water balance of the oxidation ponds and the surrounding land-disposal area is presented in Table 5 for all 16 options. Between January 1997 and October 2005, it is calculated that a total of 49.2 Mm^3 of influent water and 2.0 Mm^3 of rain water entered the oxidation ponds over the 3195 days of operation. When averaged across all options, about 73% of the treated effluent went directly into the river during 60% of the days. During the same period, approximately 21% of the effluent volume was disposed of onto the land and this occurred on about 92% of the days (Table 5). The remaining 6% of water that entered the ponds, as rainfall and return flows of effluent, matched approximately the amount of water lost as evaporation and leakage, in roughly equal amounts. On balance, the volume of effluent applied to the land matched approximately the volume of water that drained beyond a depth of 1.0 m. Rainfall and pasture evaporation losses from the land are of a comparable value. Runoff losses from the land were calculated to be very small, mainly because the disposal rules reduced application volumes whenever the soil was close to saturation.

The preferred option has been selected as Option 6. The maximum storage volume is calculated to be 339,000 m^3, which is 24,000 m^3 less than the maximum storage capacity of the current oxidation ponds (i.e. 363,000 m^3). This margin (of 24,000 m^3) is equivalent to a water depth of approximately 0.1 m, or about two days of inflow to the ponds, and is considered to provide a prudent degree of buffer capacity. Figure 3 shows the modelled rise and fall of the operating storage volume over the eight years of record used in the analysis. A tabular summary of key parameters relating to the preferred scheme (Option 6) is detailed in Table 4 below.

Table 4. The water balance of the oxidation pond and the whole Homebush, Masterton disposal site was modelled for a period of eight years of operation (1997-2006). The results presented here are for the preferred scheme (Option 6).

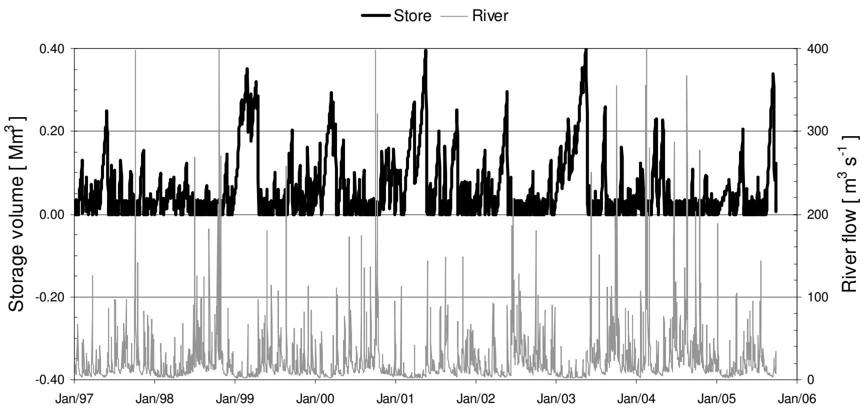
Number of days	3195	
Days to river	2017	
Days to land	2947	
Pond water balance	Volume [ml]	% inflow
Inflow	49157	100.0
Rainfall	1993	4.0
Runoff from land	50	0.1
Discharge to river	-37860	-77.1
Discharge to land	-10310	-21.0
Runoff to river	-303	-0.7
Evaporation from pond	-1465	-3.0
Leakage from pond	-1566	-3.2
Max. pond storage (Mm^3)	339	
Site water balance	Volume [ml]	% irrigation
Irrigation	10311	100.0
Rainfall	5571	54.0
Evaporation	-5235	-50.8
Drainage	-10031	-97.3
Runoff	-353	-3.5

Key points from the above table are (note that these are average figures based on the modelled eight years):

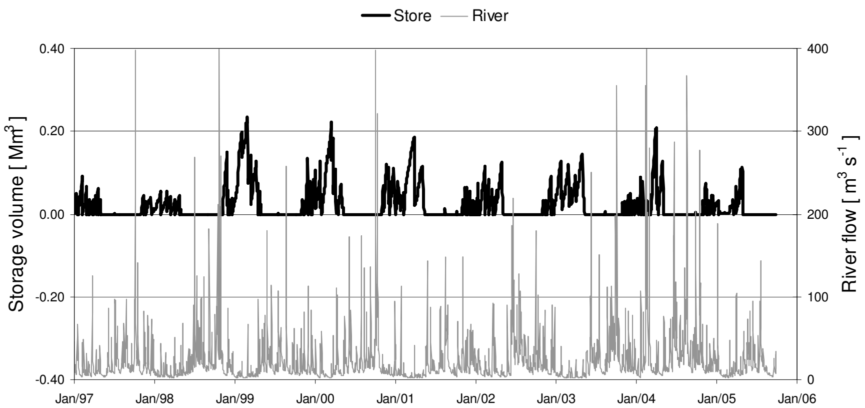
- Approximately 77% of effluent is discharged to the river
- Approximately 21% of effluent is applied as irrigation
- Discharge to the river occurs on 63% of days
- Irrigation takes place on 92% of days.

DRY-LAND FARM (CUT-AND-CARRY OPERATION)

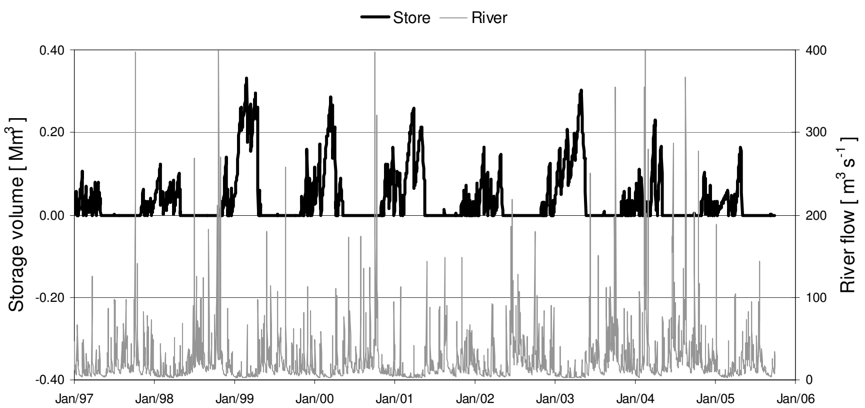
A non-irrigated option was established to represent a background level for water and nutrient fluxes. In the dry-land case, the site was given zero inputs of irrigation and fertilizer (Table 6). The annual pasture production was predicted to be very low (~5278 kg-dry matter (DM) ha^{-1} y^{-1}; Table 7), and the predicted transpiration rates were reduced because of mild water and nutrient stresses (Table 8).



3a. Option 1



3b. Option 3



3c. Option 6

Figure 3. SPASMO (Soil Plant Atmosphere System Model) was used to calculate changes in the storage volume for the Homebush, Masterton oxidation ponds (solid line) under a range of disposal options. Option 6 is the preferred option. Option 1 restricts winter disposal to the river unless the mean river flow >12.3 m^3 s^{-1}. Option 3 provides for a 50% increase in the volume of effluent disposed to land. The mean daily river flow recorded at Wardell's Bridge near the Homebush site is shown by the grey line.

Table 5. An option analysis was carried out to determine the effect of a range of disposal options on the water balance of the Homebush, Masterton oxidation ponds and the surrounding land-disposal sites (see pages 14 and 15 for a description of the options). Unless otherwise stated, all numbers refer to a total volume (Mm^3) as calculated for the period 1 January1997 to 30 October 2005. The shaded cells represent the preferred option (Option 6).

Options	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
Pond water balance																	
Effluent inflow	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2	49.2
Volume rainfall	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Runoff return to pond	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
Effluent to the river	-37.9	-37.8	-36.8	-33.3	-36.7	-36.0	-37.9	-37.3	-39.7	-37.9	-37.9	-37.9	-37.9	-37.9	-37.1	-38.6	-36.3
Effluent to the land	-10.3	-10.3	-11.3	-14.9	-11.5	-12.2	-10.3	-10.9	-8.5	-10.3	-10.3	-10.3	-10.3	-10.3	-10.3	-10.3	-11.8
Evaporation loss	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5	-1.5
Pond leakage	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-1.6	-2.3	-0.8	-1.6
Site water balance																	
Effluent to the land	10.3	10.3	11.3	14.9	11.5	12.2	10.3	10.9	8.5	10.3	10.3	10.3	10.3	10.3	10.3	10.3	11.8
Rainfall to land	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
Drain from land	-10.0	-10.0	-11.0	-14.6	-11.2	-11.9	-10.0	-10.6	-8.2	-10.0	-10.0	-10.0	-10.0	-10.0	-10.0	-10.0	-11.6
Evapotranspiration from land	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2	-5.2
Runoff losses	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4	-0.3	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4	-0.4
Operational aspects																	
Maximum storage volume	0.352	0.416	0.335	0.234	0.331	0.299	0.339	0.342	0.411	0.352	0.264	0.363	0.408	0.507	0.335	0.369	0.284
% time to river	58	23	58	57	58	57	63	58	58	58	61	57	57	56	58	58	57
% time to land	92	92	91	98	91	92	92	92	92	92	92	92	92	92	92	92	92

In the absence of irrigation, drainage losses were predicted to be quite low (~229 mm y^{-1}). This represents about 25% of the average annual rainfall (940 mm y^{-1}). Because of such low pasture growth, the cut-and-carry operation from a dry-land farm, with no supplemental nitrogen or phosphorus fertilizer, would remove just 154 kg ha^{-1} of nitrogen and just 5 kg ha^{-1} of phosphorus each year (Table 7). Under the dry-land farm the average nitrate-nitrogen concentration in drainage water (at a soil depth of 1.0 m) is predicted to be about 6.0 mg L^{-1}, on average (Table 8), or about half the New Zealand drinking water standard. Annual nitrate leaching losses under the dry-land farm are calculated to be just 15 kg ha^{-1}, on average, and the corresponding P losses are expected to be negligible in the short term (<20 years).

IRRIGATED FARM (PREFERRED OPTION WITH CUT-AND-CARRY PASTURE)

Increased hydraulic and nutrient loading to the land will inevitably have an impact on pasture growth and affect the quality of groundwater under the disposal site. Under the preferred option, some 2013 mm of effluent water will be irrigated onto the pasture each year, when averaged across the whole pasture site. This amount of water greatly exceeds pasture requirements for transpiration and results in a large amount of drainage. The calculations show the annual drainage losses will approximately match the amount of effluent water being applied to the land surface (Table 6).

Annual pasture production from the irrigated farm will increase to 11.1 T ha^{-1}/annum of DM, on average, partly because of the extra 231 kg ha^{-1} of nitrogen that is contained in the irrigation water (Table 6). This compares with 5.3 T/ha/annum of DM without irrigation and adequate nitrogen fertilizer (Table 7). Nonetheless, pasture production will still be less than optimum because of nitrogen limitations under the cut-and-carry operation. Harvesting the grass will remove some 303 kg ha^{-1} of nitrogen from the site each year. The corresponding removal of phosphorus will amount to about 23.6 kg ha^{-1} each year, on average. In addition, the average nitrate nitrogen concentration of the drainage water under the irrigated farm would be just 2.7 mg L^{-1}. This is about half the concentration of drainage water that quits the root-zone of the dry-land pasture. Effluent irrigation will not only increase pasture growth and nutrient uptake, it will also lead to a reduction in the nitrate concentration of the drainage water due to a dilution effect. The higher nitrate values predicted under the dry-land farm are a consequence of low pasture uptake and very low leaching losses.

Table 6. Average hydraulic and contaminant loading for the border-strip areas in the Homebush, Masterton site (Note: The annual balances for all sites are shown in Tables D1-D10 of Appendix D). N = nitrogen; P = phosphorus.

Option	Transpiration (mm y^{-1})	Irrigation (mm y^{-1})	Drainage (mm y^{-1})	N added (kg-N ha^{-1} y^{-1})	P added (kg-N ha^{-1} y^{-1})	Bacteria (cfu m^{-2} y^{-1})
Dry-land	618	0	229	0	0	0
Irrigated	814	2014	1999	231	50.3	1.05×10^7

Table 7. Average pasture production (dry matter, DM = kg DM ha^{-1} y^{-1}), net nitrogen fixation (N fixed) and the corresponding amount of nitrogen (N-cut) and phosphorus (P-cut) removed from the border strips in the Homebush, Masterton site via a cut-and-carry operation. (Note: The annual balances for all sites are shown in Tables D1-D10 of Appendix D).

Option	DM (kg ha^{-1} y^{-1})	N fixed (kg ha^{-1} y^{-1})	N cut (kg ha^{-1} y^{-1})	P cut (kg ha^{-1} y^{-1})
Dry-land	5278	112	154	5
Irrigated	11059	171	303	23.5

Table 8. Average leaching losses and contaminant concentrations at the Homebush, Masterton site at a depth of 1.0 m. The results have been averaged for the ten border strip areas. (Note: The annual balances for all sites are shown in Tables D1-D10 of Appendix D).

Option	NO_3^- loss (kg ha^{-1})	NH_4 loss (kg ha^{-1})	P loss (kg ha^{-1})	Bacteria (cfu m^{-2})	NO_3^- (mg-N L^{-1})	NH_4^+ (mg-N L^{-1})	P conc. (µg-P L^{-1})	Bacteria (cfu/0.1L)
Dry-land	13.3	1.6	0.0	-	6.0	0.7	4.7	0.0
Irrigated	56.7	6.5	-	1.86×10^5	2.7	0.25	250.	4.1

PLANT UPTAKE OF NUTRIENTS

The free-draining soils (Zones 1 to 10) will be planted in a perennial ryegrass and clover sward. Such pastures are normally high yielding (15-20 t dry matter (DM) ha^{-1} y^{-1}) on fertile sites, and they have the potential to remove large amounts of nutrient annually (500–600 kg ha^{-1} of N, 130–160 kg ha^{-1} of P, 140–160 kg ha^{-1} of potassium (K)) under a cut-and-carry operation (Morton et al. 2000). The poorly drained soils (Zone 11) will be planted with short-rotation forest of *Eucalyptus ovata* trees that will be coppiced on a four-year rotation (John Lavery, Forest Research, pers. comm.). Yields from short-rotation tree crops can vary greatly under New Zealand conditions. However, under municipal effluent irrigation *Eucalyptus* trees have been reported to yield over 25 oven-dry tonnes of total biomass per hectare per year (Nicholas et al. 2000) and to remove up to 200 kg ha^{-1} y^{-1} of N when coppiced on a 3-4 year cycle (Barton et al. 1991).

A modelling approach is used here to calculate crop growth and nutrient uptake. Details are given in Appendix A. Table 9 presents average values of the predicted nutrient budget for two pastures (low and high irrigation volumes), as well as the short-rotation forest. Under the current design rules for operation, different amounts of effluent will be disposed onto the land area, at a rate up to but not exceeding the maximum, depending on the soils' hydraulic capacity to store and drain the water. Sites receiving more effluent will be more productive, all other factors being equal. On the more free-draining soils, the hydraulic loads will be higher and water will be non-limiting at all times. However, pasture and tree production may still be comparatively low because of the poor nitrogen and phosphorus status of the soils (Moir et al. 2000).

The annual pasture production on individual irrigation zones is calculated to be between 9.3-12.2 t DM ha^{-1} under the preferred disposal scheme (Table D7). The lowest productivity is from those zones receiving the least amount of treated effluent, as reported for Zone 3 (Table 9). Some 300-415 kg ha^{-1} of nitrogen will be taken up by the pasture. This includes an annual nitrogen fixation of between 156-180 kg ha^{-1}, based on 15% clover content. The cut-and-carry operation will remove most of the pasture DM (between 75-85%) and nutrients from the site. However, some pasture residuals (DM > 1.0 T ha^{-1}) will be topped in the autumn and left to decompose over the winter time. Low pasture production is partly because the amount of nitrogen added from the effluent (130-290 kg N ha^{-1} each year) is less than the pasture needs for optimum growth.

Table 9. Estimated nutrient budget for 'cut-and-carry' pasture (Zone 3 has a low irrigation loading and Zone 7 has a high irrigation loading) and a short-rotation *Eucalyptus ovata* forest (Zone 11) at the Homebush, Masterton site. Unless otherwise stated, all units are kg ha^{-1} y^{-1}. These results are for the preferred option (Option 6). Balance calculations for other sites and other options are presented in Appendix D, Tables D1-D8.

Component		Zone 3	Zone 7	Zone 11
Inputs	Hydraulic load [mm y^{-1}]	1142	2509	788
	Nitrogen (N) effluent	131.4	288.4	90.8
	N fixed by clover	156.8	180.1	-
	Phosphorus (P) effluent	28.1	63.0	19.7
Outputs	Dry matter harvest	9267	12151	12353
	N harvest	297.8	415.6	90.5
	P harvest	19.6	34.6	4.5

The irrigation adds between 28 and 63 kg P ha^{-1} each year to the soil. Some 20-35 kg ha^{-1} of phosphorus will be taken up each year by the pasture. This will be subsequently harvested and most of it (80-90%) will be removed from the site under the cut-and-carry operation. Corresponding values of nutrient uptake by the short-rotation forest are predicted to be about 91 kg ha^{-1} of nitrogen and about 5 kg ha^{-1} of phosphorus each year. The model calculates comparatively low rates of tree growth partly because of the low nitrogen status of the soils. Trees are able to take up the equivalent of all the applied nitrogen, and about 25% of the applied phosphorus.

TREATMENT CAPACITY OF THE SOILS

The soil acts as a reservoir either to filter, to retain or to remove particular constituents from the effluent. The degree of renovation of the effluent will depend on the interaction between soil processes and water movement. The main potential for adverse outcomes relates to pathogens (bacteria), and nutrients (phosphorus and nitrogen) leaching from the base of the root-zone to the receiving waters. To be effective, the effluent needs both a sufficient residence time and adequate travel distance in the soil to adsorb nutrients and attenuate contaminants and bacteria. For the next section of this report, we have used a longer daily time series to simulate the treatment capacity of the soils. We have used a period of 28 years that represents the maximum length of available climate and river flow records. Unfortunately, the corresponding time series of influent volumes was not available. So, for the purpose of calculation, the missing influent volumes were synthesised by assuming a steady base flow (some 8500 m^3 d^{-1}) combined with daily fluctuations (0 to ~30000 m^3 d^{-1}) determined from the sequence of rainfall totals during the previous month (data not shown). A least-squares regression approach was used to determine appropriate weighting factors needed to generate an extended time series that had similar statistical properties to the existing 8 years of records from the treatment plant.

NITROGEN

Total nitrogen content of the treated effluent from the oxidation ponds is expected to be about 11.5 mg L^{-1}, on average. For the purpose of modelling, we have assumed 50% of the applied mineral nitrogen is in the form of ammonium, and the remainder is in the form of nitrate that is in solution. Normally ammonium adsorbs onto the soil's mineral and organic matter and so its downward movement through the soil profile is retarded relative to the drainage water. However, ammonium is also rapidly oxidised (half life is typically 2-10 days) by microbial processes to nitrate, which is highly mobile and will travel freely though the soil, being transported downwards along with the percolating drainage water.

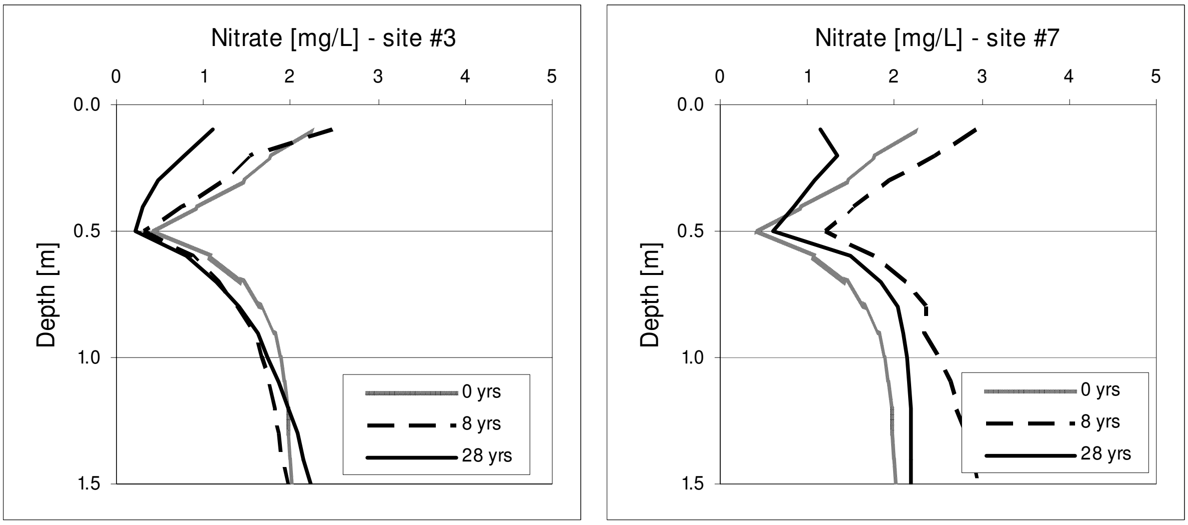


Figure 4. Profile of nitrate-nitrogen in soil water draining through disposal site #3 (annual loading = 130 kg nitrogen (N) ha^{-1}) and #7 (annual loading = 290 kg N ha^{-1}) at the Homebush, Masterton site. The New Zealand Drinking Water Standard for nitrate-N is 11.3 mg L^{-1}. These results are for the preferred option (Option 6).

Model calculations reveal nitrate leaching to be of little concern with regard to potential contamination of the groundwater as a result of effluent application. The solution concentration in drainage water at 1 m depth is predicted to remain well below the New Zealand Drinking Water Standard of 11.3 mg L^{-1} of nitrogen, even after a period of 28 years (Figure 4). Pasture roots are largely confined to the top 0.5 m of soil. Nitrogen uptake by pasture results in a local minimum for nitrate near the base of the root zone. There is a subsequent small increase in nitrate concentrations beyond 0.5 m, and this reflects nutrients that quit the base of the root zone. There is unlikely to be any significant accumulation of nitrate in the soil profile over time. This is because nitrogen uptake by the pasture can easily account for all the applied nitrogen. Furthermore, the cut-and-carry process for pasture will remove a large fraction of the pasture nitrogen (80-90%) from the site, thereby creating little excess nitrogen that can leach.

PHOSPHORUS

Phosphorus is a relatively immobile element in most New Zealand soils. When applied to land, it will normally be bound to the soil and accumulate within the top 10-20 cm of the root-zone, where it can be taken up by plants. The total phosphorus content of treated effluent from the oxidation ponds is expected to be 3.2 mg L^{-1}, on average. Most of this phosphorus will be in the form of dissolved reactive phosphorus (DRP) which is readily taken up by plants, yet strongly adsorbed to the soil's mineral and organic surfaces. In order to model P transport through the soil, a series of equilibrium isotherm experiments were measured to determine soils' capacity to transport and retain phosphorus. Appendix B presents the approach that was used to determine a relationship between P transport properties and the local soil texture (Tables B1-3).

Analysis shows that the P retention capacity on the Homebush soils is very low (8-19%). This means, over time, the surface soil could eventually become "saturated" with phosphorus, under a heavy load, thereby possibly enabling some leaching of P to occur. In addition, drainage via macropore flow could further enhance the downward movement of phosphorus, although the deeper clay-rich layers would tend to retain such phosphorus and thereby limit leaching losses into the groundwater. Modelling was carried out to assess the environmental fate of the surface-applied phosphorus. Figure 5 illustrates the predicted situation for two Zones receiving different amounts of effluent. Zone 3 receives the lowest effluent input, equivalent to about 28 kg P ha^{-1} each year, while site 7 receives the highest effluent input, with some 63 kg ha^{-1} of phosphorus being added each year.

Following 28 years of effluent application, some 61% of the phosphorus applied to Zone 3 will remain in the top 1 m of soil. While the soil concentration slowly increases over time, it is still a factor of 2-6 times lower than the maximum concentration at saturation (Figure 5 cf. Q values in Table B3). The soil solution concentration at a depth of 1 m is predicted to remain below 0.02 mg L^{-1}. Over the 28-year period, less than 8 kg P ha^{-1} is expected to leach below 1 m. The potential impact of P leaching on groundwater quality under zone 3 is expected to be negligible.

In contrast, Zone 7 receives almost twice the amount of P each year. The top soil under Zone 7 has a much lower P retention capacity because of its lower clay content. This means a greater fraction of the surface-applied P will move downward through the soil profile (cf. top and bottom panels of Figure 5). Model calculations over a 28-year period indicate some 66% of the applied phosphorus will still remain in the top 1 m of soil. Although the solution concentration entering the ground water could slowly rise to 0.2 mg L^{-1}, additional dilution in the ground water, combined with strong adsorption by the deep clay-rich layers means the off-site impacts on surrounding ground water are still likely to be negligible.

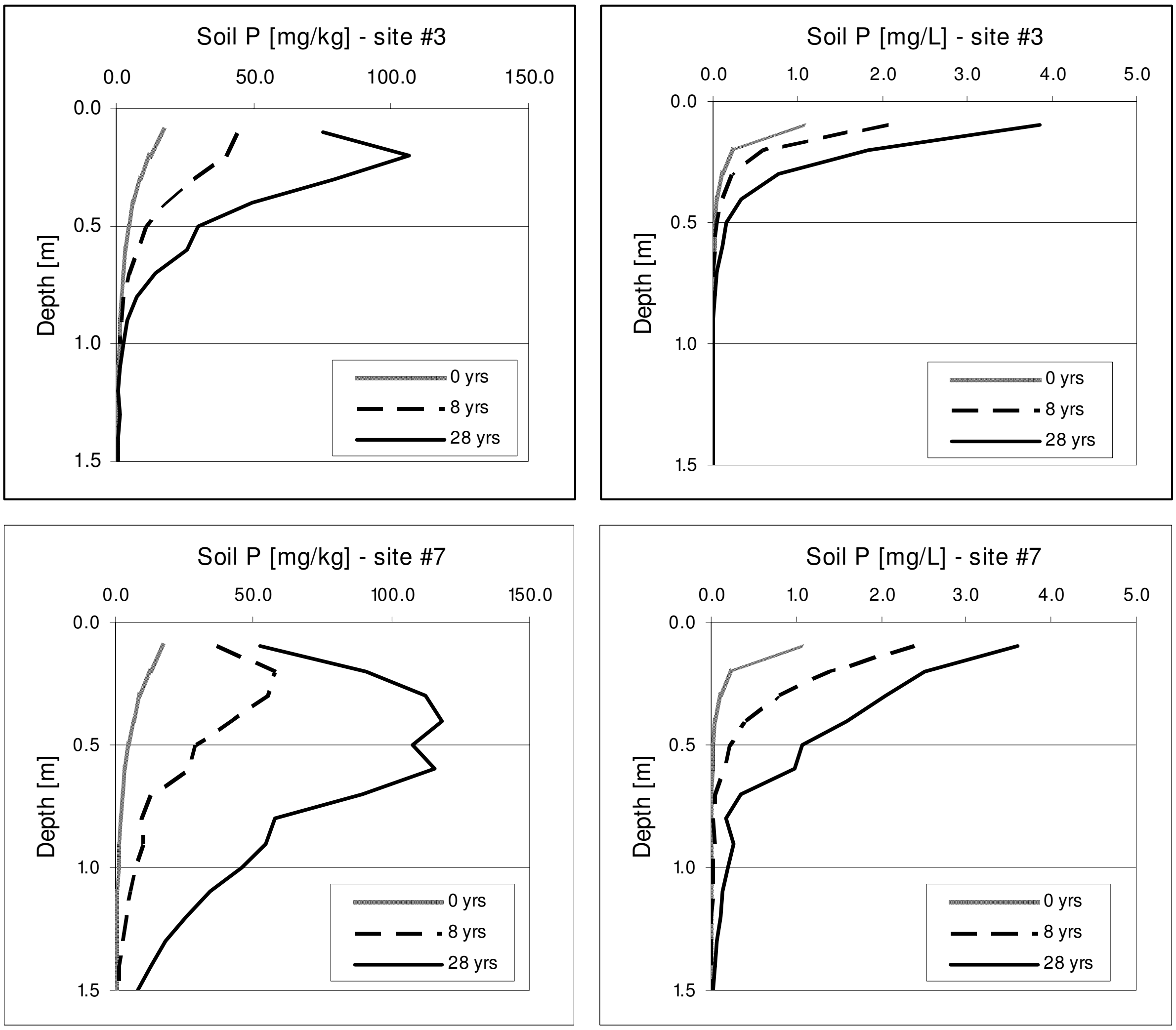


Figure 5. The concentration of phosphorus attached to soil (left panel) and the corresponding solution concentration (right panel) in water that that drains under disposal sites 3 and 7 at the Homebush, Masterton site. The annual loading equates to 61 kg ha^{-1} of phosphorus, applied at an influent concentration of 2.5 mg L^{-1}. Zone 7 is loamy silt on a sandy loam. The results are for the preferred option (Option 6).

Irrigation of treated effluent adds more phosphorus to the soil than can be utilized by the pasture. As a result, phosphorus slowly accumulates in the top 1.0 m of the soil profile. Following 28 years of application, the maximum concentration of soil P (inorganic form) is calculated to reach about 100-125 mg kg^{-1}. This is higher than is needed to achieve optimum pasture growth, all other factors being non-limiting (Moir et al. 2000), yet unlikely to cause any problems with respect to pasture production or soil function. The maximum sorption capacity of the clay rich layers is between 410-615 mg kg^{-1} (Table B3). Even after 28 years of irrigation, it is calculated that the maximum soil P concentration should remain <25% of phosphorus sorption saturation (PSS). In the Netherlands, a PSS of 25% has been established as the critical point above which the potential for P movement into groundwater becomes unacceptable, while in Belgium a limit of 30% has been assumed (De Smet et al. 1996).

There are generally no adverse effects of high soil P concentrations on plant growth and soil function. Reports from the USA, where high rates of manure and effluent irrigation are regularly applied to grassland, show no loss in productivity in soils where the available P concentrations exceed 180 mg kg^{-1} (Johnson et al. 2004). Walen and Chang (2001) reported increased crop production under cultivated soils receiving long-term annual manure applications, with available soil P concentrations exceeding 800 mg kg^{-1}. This is some 2-8 times higher than we predict for the Homebush site. We conclude that the accumulation of P in the top soil is unlikely to cause any long-term problems with pasture production or soil function at Homebush.

BACTERIA

Sewage effluent contains a variety of pathogens, including bacteria (e.g., *Salmonella* spp., *Campylobacter* spp.) and viruses (e.g., *Rotavirus, Norwalk virus*) which can cause disease in humans and livestock. Thus, land application of effluent may increase the risk of groundwater contamination by these pathogenic micro-organisms. The New Zealand Drinking Water Standard for bacteria currently set at one colony forming unit (cfu) per 100 ml (MFE 2005). Groundwater concentrations exceeding this guideline value are indicative that faecal matter, and possibly other disease-causing organisms (pathogens), may be present.

Modelling was carried out to assess the environmental risks of bacteria reaching the groundwater. The two main processes that remove bacteria from the soil are colloidal filtration and inactivation. Colloidal filtration describes how bacteria are intercepted by and ‘stick’ to the soil particles. Inactivation describes the rate of bacteria ‘die off’ under natural conditions. These processes are modelled in a standard manner using parameters values taken from the literature (Yates and Ouyang 1992). The calculations also account for advection, storage, dispersion, and adsorption of bacteria as it moves through the soil profile. More details of the bacteria transport component of the model are presented in Appendix A. For this study we use *Escherichia coli* as an indicator of faecal pollution because it is found exclusively in the intestinal tract of humans (and it is present in high numbers in effluent). Furthermore, *E. coli* does not usually multiply in the environment (Geldreich 1996), so the number predicted can be interpreted quantitatively.

The average E. *coli* concentration of the treated effluent from the oxidation ponds is assumed to be 1000 cfu per 100 ml in the winter and 200 cfu per 100 ml in the summer. It is calculated that between 95-99% of surface-applied *E. coli* are removed during transport through the top 1 m of soil. Inactivation is predicted to account for almost all the applied bacteria (Tables D1-D9 of Appendix D), while colloidal filtration acts to retard the downward movement by trapping (sticking) *E. coli* to the soil’s clay surfaces. Leaching to groundwater accounts for the remainder. None of the simulations predicts a significant accumulation of bacteria numbers over time (Figure 6). The average concentration in the drainage water at a depth of 1 m sometimes reaches a concentration in the order of 15 cfu per 100 ml, which exceeds New Zealand Drinking Water Standards by a factor of up to 15 times. *E. coli* can survive for up to six weeks in freshwater (Filip et al. 1987; Edberg et al. 2000). However, additional die off and dilution in the groundwater is expected to reduce these concentrations further. It is concluded that *E. coli* in treated effluent added to land is unlikely to have a detrimental impact on the quality of the groundwater.

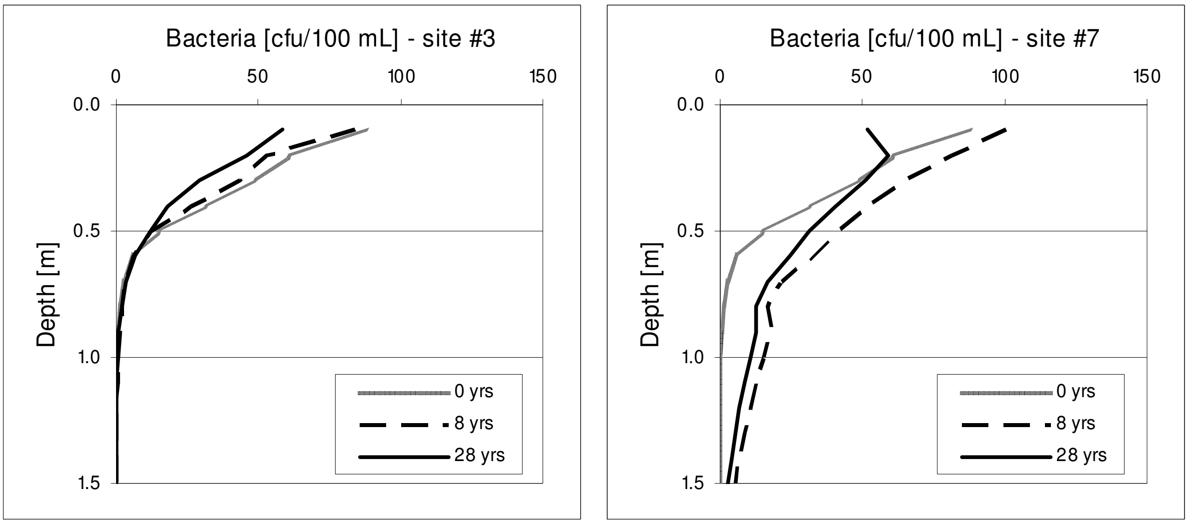


Figure 6. Profiles of the soil bacteria (*Escherichia coli* solution concentration) at disposal site #3 (low hydraulic load) and #7 (a high hydraulic load) receiving effluent from the Masterton treatment plant. The results relate to the preferred option (Option 6).

LONG-TERM SUSTAINABILITY OF SOILS

A key issue in relation to effluent disposal by irrigation is whether the proposed irrigation rate is sustainable in terms of soil conditions, ground water levels and ground water quality. The main potential for adverse outcomes relate to pathogens, and nutrients (phosphorus and nitrogen) contaminating ground and surface waters. Excessive volumes of irrigation water could also degrade soil structure by blocking pores, limiting aeration and reducing soil permeability to water. Such effects could eventually reduce crop growth, or lead to an hydraulic failure of the land-based disposal system.

EFFECTS ON SOIL STRUCTURE

One issue that can be of concern when land is irrigated by effluent is the application and impacts of salts. These concerns relate to the effects of total soluble salts on plant health, and the effects of sodium on soil structure. Salt accumulation in soil can reduce crop production, while sodium accumulation can degrade soil structure thereby reducing the permeability for water. Winter rainfall and flushing of salts will generally limit any damage.

Generally, the risk of salt or sodium problems for New Zealand soils irrigated with sewage effluent is considered to be low (Tipler 2000). The electrical conductivity of wastewater is typically 2.5 dS m^{-1} and the sodium absorption ratio (SAR), which is a key parameter used to assess the risk of adverse soil effects due to sodium loadings, lies between 4 and 7. These levels are generally acceptable for irrigation water. Nevertheless, periodic monitoring of the soil exchange balance is recommended, given potential effects on crop nutrient balance.

6. SUMMARY

Modelling has been carried out to determine both the storage requirements of the oxidation ponds and the environmental footprint from the land-based disposal. The calculations utilize a complete record of eight years of climate, river flow and influent data that are required as input to the SPASMO model. Initially a base-case model was established. Then a range of different options was tested in order to assess the impact of altering various operational parameters. The outcome of the option testing was that Option 6 was selected as the preferred scheme. The basis for selection of Option 6 is that the selected discharge rules will provide a strong environmental enhancement, and also achieve a satisfactory storage volume. Key impacts for the preferred scheme are summarized below.

DISCHARGE RULES

- Irrigation will occur whenever possible
- Discharge direct to river will occur:
  - In summer only when the river flow is greater than the median river flow
  - In winter only when the river flow is greater than the half median river flow
- Whenever there is a direct discharge to the river, the ratio of river flow to effluent flow will be at least 30 to one, up to the maximum effluent flow
- The maximum effluent flow for a direct discharge is 104,000 m^3 d^{-1} (1200 L s^{-1}).

STORAGE VOLUME

The maximum operating storage volume is 339,000 m^3, which is 24,000 m^3 less than the maximum storage capacity of the current oxidation ponds (i.e. 363,000 m^3). This margin (of 24,000 m^3) is equivalent to a water depth of approximately 0.1 m, or about two days of inflow to the ponds, and is considered to provide a prudent degree of buffer capacity.

DISPOSAL OF EFFLUENT UNDER THE PREFERRED SCHEME

- Over the eight years of record, disposal by irrigation occurs on 92% of days with approximately 21% of the inflow volume being disposed of by irrigation
- Over the eight years of record, a direct discharge to river occurs on approximately 63% of days, with approximately 77% of the inflow volume being discharged directly to the river.

IMPACT OF EFFLUENT USED AS IRRIGATION

The fate of nitrogen, phosphorus and bacteria (*E. coli*) in the upper soil layers was assessed as part of the model development. An estimate was made of the quantity and quality of the water that would leach to the underlying groundwater, with further work on the assessment of the impact on groundwater quality being undertaken and reported separately by Pattle Delamore Partners.

Key conclusions with respect to these parameters are summarised below.

Nitrogen
Modelling shows leaching of nitrate nitrogen to be of little concern.

- Each year, on average, the effluent irrigation adds about 231 kg N ha^{-1} to the pasture zones and 90 kg N ha^{-1} to the short-rotation forest
- The annual nitrogen uptake by pasture is calculated to be between 300-415 kg N ha^{-1} and this include an annual nitrogen fixation of between 156-180 kg N ha^{-1} based on a 15% clover content. The corresponding value of nitrogen uptake by the short-rotation forest is calculated to be about 90 kg N ha^{-1}
- The cut-and-carry operation will remove most of the pasture dry matter (between 75-85%) and nutrients from the site
- The average nitrate-nitrogen concentration of drainage water at 1.0 m is calculated to be 2.7 mg L^{-1}. This is about four times lower than the current New Zealand Drinking Water Standard (MFE 2005) and therefore should not pose a significant threat to the quality of the groundwater.

Phosphorus
Phosphorus accumulation and leaching has been assessed for a 28-year period, with the results showing:

- Each year, on average, the effluent irrigation adds about 50 kg P ha^{-1} to the pasture zones and 20 kg P ha^{-1} to the short-rotation forest
- The annual uptake of P by pasture is calculated to be between 20-35 kg N ha^{-1} for the pasture zones and about 5 kg P ha^{-1} for the short-rotation forest. Thus, the pasture takes up about half the applied phosphorus, which is largely removed from the site, while the trees take up about 25%
- Following 28 years of effluent irrigation, it is calculated that the maximum soil P concentration will remain <25% of the phosphorus sorption capacity. Although the solution concentration entering the ground water could slowly rise to 0.2 mg L^{-1}, additional dilution in the ground water, combined with strong adsorption by the deep clay-rich layers, means the off-site impacts on surrounding ground water are likely to be negligible.

Bacteria (E. coli)
Bacterial transport through the soil has been assessed for a 28-year period, with the results showing:

- Between 95-99% of surface-applied *E. coli* are removed during transport through the top 1 m of soil. Inactivation (die off) will account for almost all the applied bacteria
- None of the simulations predicts a significant accumulation of bacteria numbers over time. The average concentration in the drainage water at a depth of 1 m sometimes exceeds New Zealand Drinking Water Standards by a factor of 1-15. However, additional die off and dilution in the groundwater is expected to reduce these concentrations further
- It is concluded that *E. coli* in treated effluent added to land is unlikely to have a detrimental impact on the quality of the groundwater under the disposal site.

The modelling has shown that the land area provided for disposal is able to accommodate the water and nutrient loads being applied and provide an effective filtering capacity. However, three conditions will need to be taken into account:

- The hydraulic capacity of the soils should not degrade under irrigation

- The river flow characteristics should remain similar to past years (i.e. there should be no severe and prolonged droughts that alter the duration of low flows in summer)
- If effluent volumes and composition change compared with past years, then the model outputs will also change.

Predicting how these characteristics might change over time is beyond the scope of this desk-top modelling exercise, although this could be considered in the future.

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APPENDIX A: MODEL DESCRIPTION

A.1. A general description of the SPASMO model

The SPASMO computer model considers water, solute (e.g. nitrogen and phosphorus), and microbial (e.g. viruses and bacteria) transport through a 1-dimensional soil profile. The soil water balance is calculated by considering the inputs (rainfall and irrigation) and losses (plant uptake, evaporation, runoff and drainage) of water from the soil profile. The model includes components to predict the carbon, nitrogen and phosphorus budget of the soil. These components allow for a calculation of plant growth and uptake of both N and P, various exchange and transformation processes that occur in the soil and aerial environment, recycling of nutrients and organic material to the soil biomass, and the addition of surface-applied fertilizer and/or effluent to the land. The filtering capacity of the soil with regard to micro-organisms is modelled using an attachment-detachment model with inactivation (i.e. die-off) of microbes.

Model results for the water balance are expressed in terms of mm (= one litre of water per square metre of ground area). The concentration and leaching losses of nutrients are expressed in terms of mg L^{-1} and kg ha^{-1}, respectively. The microbial concentrations and leaching losses are expressed in terms of colony forming units, cfu L^{-1} and cfu m^{-2}, respectively. All calculations run on a daily basis and the results are presented at the paddock scale.

A.2. Water and solute flow through the soil

The flow of water through the soil profile is simulated using a capacity model similar to that of Hutson & Wagenet (1993), in which the soil water is divided into mobile and immobile phases. The mobile domain is used to represent the soil's macropores (e.g. old root channels, worm holes and cracks) and the immobile domain represents the soil matrix. The equations describing water and contaminant flow are simple, but lengthy, and so they are not repeated here (see Hutson & Wagenet (1993) for details).

On days when there is rain or irrigation, both applied water and any dissolved solutes are added to the surface layer. The maximum amount of water that can infiltrate the soil is limited by the storage capacity of the profile, and the minimum saturated hydraulic conductivity of the subsoil. The water content of topsoil (0-30 cm) can't exceed saturation, otherwise some runoff is generated. After rainfall or irrigation, water is allowed to percolate through the soil profile, but only when the soil is above field capacity. The infiltrating water first fills up the immobile domain and, once this domain is filled, it then refills the mobile domain as the water travels progressively downward through the soil profile. If the soil is above field capacity, then the infiltrating water and solute resides in the mobile domain where it can percolate rapidly down through the soil profile until it reaches a depth where the water content is no longer above field capacity. This macropore flow is rapid and it does not allow enough time for exchange between the mobile and immobile domains. As a consequence, the two flow domains are temporarily at quite different solution concentrations as water percolates through the soil profile.

Subsequently, on days when there is no significant rainfall, there is a slow approach to equilibrium between the mobile and immobile phases, driven by a difference in water content between the two domains. The rules for the subsequent slow approach to equilibrium between the mobile and immobile phases within a depth, or model segment, are described in their original scientific paper (Hutson & Wagenet 1993).

If a soil layer is below field capacity or if there is no rainfall or irrigation to generate percolation, then each soil segment i is brought towards equilibrium with the segment i+1 beneath, starting from the top of the profile. This redistribution of water is achieved by (i) calculating the amount of water required to move upwards or downwards so that each soil segment reaches an equilibrium water potential with its neighbour, and (ii) allowing only half this water to move, together with its dissolved solute. After all segments have been adjusted, each solute (i.e. ammonium, nitrate, phosphorus and bacteria) is repartitioned between aqueous and solid (sorbed) phases, assuming complete equilibrium between mobile and immobile phases.

The total water content in each soil segment, W_T [mm], is given by the sum of the water contents in the immobile and mobile soil domains

$$W_T = W_I + W_M \,. \qquad \text{[Eq. A1]}$$

and total amount of solute in each soil segment, M_C [mg m^{-2}], is calculated as

$$M_C = C_I W_I + C_M W_M + S\,\rho\,\Delta z \qquad \text{[Eq. A2]}$$

Here, C is the solution concentration [mg L^{-1}], S represents the amount of sorbed solute [mg kg^{-1}], ρ is the bulk density [kg L^{-1}], Δz is the segment thickness [mm], and the subscripts I and M refer to the immobile and mobile domains, respectively. The sorption of ammonium and nitrate is described using a simple linear isotherm of the form

$$S = K_D C \qquad \text{[Eq. A3]}$$

where K_D represents the distribution coefficient [L kg^{-1}]. In the case of nitrate, which is considered to be inert, we assume no adsorption and set K_D equal to zero. The equilibrium solution concentration, C, in both mobile and immobile phases of nitrate and ammonium, is then calculated as

$$C = M_C / \left(\rho\, S\, \Delta z + W_T\right). \qquad \text{[Eq. A4]}$$

The sorption of phosphorus is non-linear and is described using a Langmuir isotherm of the form

$$S = \frac{Q\,bC}{1 + bC} \qquad \text{[Eq. A5]}$$

where Q is the maximum total mass of phosphorus at saturation per mass unit of dry soil [μg g^{-1}], and b is an empirical constant, with units of inverse of solution concentration [L mg^{-1}]. The b-parameter defines the point where the soil is at half-saturation with respect maximum sorption of P.

Bacterial transport is calculated using the same convection-dispersion type equation as for water and solute transport, with additional terms used to represent the kinetic sorption of bacteria to soil's mineral particles as well as the subsequent detachment and transfer of bacteria between the aqueous and solid phases. The attachment-detachment process is described using first-order rate constants that strongly depend on soil water content (Logan et al. 1995). The rate of change in the solid-phase is modelled as

$$\rho \frac{\partial S}{\partial t} = \theta k_a \psi C - k_d \rho S \qquad \text{[Eq. A6]}$$

Here k_a is the first-order deposition (attachment) coefficient [d^{-1}], k_d is the first-order entrainment (detachment) coefficient [d^{-1}], and ψ is a dimensionless colloid retention function [-] that describes blocking of the sorption sites. This ψ-factor is calculated from the size of the sand grains and the relative solid-phase concentration (Johnson & Elimelich 1995). The attachment coefficient is calculated using a quasi-empirical formulation that takes account of the mean grain diameter of the porous media d_c [mm] and the pore-water velocity υ [mm d^{-1}], as well as terms to describe the collector efficiency η [-], and the collision (or sticking) efficiency α [-]

$$k_a = \frac{3(1-\theta)}{2 d_c} \eta \alpha \upsilon \qquad \text{[Eq. A7]}$$

The mathematical formulation of these terms, and suggested parameter values are given in Simunek et al. (2005). The collector efficiency accounts for the combined effects of particle size (e.g. bacteria or virus), fluid density and viscosity, pore-water velocity, and the water content and temperature of the soil. Because attachment is (approximately) inversely proportional to the grain size of the soil particles, finer grained soils such as silts and clays tend to be more efficient at trapping bacteria that are transported with the drainage waters. Furthermore, the smaller sized microbes (i.e. virus cf. bacteria) are less likely to be intercepted by the soil particles (i.e. have a smaller collector efficiency), so the relative value of k_a is reduced. For the purpose of modelling, the ratio k_a/k_d has been set to a constant value of 100. Other parameters used in modelling bacterial (i.e. *E. coli*) transport through the soil are discussed in Appendix B.

A.3. Calculation of crop water use

The proposed site at Homebush comprises a combination of cut-and-carry pasture and a separate area of *Eucalyptus ovata* trees. A standard crop-factor approach is used to relate crop water use to the prevailing weather and physiological time of development. The procedure is based on guidelines given by the Food and Agriculture Administration (FAO) of the United Nations (Allen et al. 1998). Daily values of global radiation, air temperature, relative humidity and wind speed are required for the calculation. These have been downloaded from the NIWA database using historical records from Te Ore Ore (Climate Station No. 7578) near Masterton. The reference evaporation rate, ET_0 [mm d^{-1}], is calculated as

$$\lambda ET_0 = \frac{s(R_N - G_H) + \rho_A c_P (e_s - e_a)/r_A}{s + \gamma(1 + r_S / r_A)} \qquad \text{[Eq. A8]}$$

where R_N [MJ m^{-2} d^{-1}] is the net radiation, G_H [MJ m^{-2} d^{-1}] is the ground heat flux, T [^oC] is the mean air temperature, e_s [kPa] is the saturation vapour pressure at the mean air temperature, e_a [kPa] is the mean actual vapour pressure of the air, s [Pa $^oC^{-1}$] is the slope of the saturation vapour-pressure versus temperature curve, γ [66.1 Pa] is the psychometric constant, and λ [2.45 MJ kg^{-1}] is the latent heat of vaporisation for water, and the terms r_S and r_A refer to the (bulk) surface and aerodynamic resistances, respectively. The surface resistance for evaporation from the pasture is set equal to 70 s m^{-1} (Allen et al. 1998). Similarly, the surface resistance for evaporation from the pond is set equal to zero. The aerodynamic resistance for both the pasture and the pond has been set equal to $208/U_2$, where U_2 is the median wind speed at a height of 2 m.

ET_0 defines the potential rate of evaporation from an extensive surface of green grass cover, of a short, uniform height, that is actively growing, completely shading the ground, and not short of water or nutrients. The potential water use of other crops, such as *Eucalyptus* trees, is then calculated

$$ET_C = K_C\, ET_0 \qquad \text{[Eq. A9]}$$

using a crop factor K_C derived from the amount of light intercepted by the leaf canopy. Light interception is a function of the leaf-area index, LAI [m^2 of leaf per m^2 of ground area] (Green et al. 2003a), and this is re-calculated each day. Coppicing the trees will reduce LAI and this impact on ET_C via a reduction in K_C.

When soil water and nutrients are non-limiting, water is extracted easily by the plant roots and transpiration proceeds at the potential rate ET_C. However, as the soil dries, water becomes more strongly bound by capillary and absorptive forces to the soil matrix. Plant roots then have to work much harder to extract water from 'dry' soil. Plants will tolerate a certain level of water deficit in their root-zone soil, yet they will eventually exhibit symptoms of water stress (i.e. reduced transpiration and loss of turgor) if the soil water content drops below a certain threshold value.

An empirical adjustment factor K_R [-] is used to represent the plant's tolerance to water stress. The total-available water TAW [mm], as defined by the difference between the water content at field capacity (-10 kPa matric potential) W_{FC} [mm] and wilting point (-1500 kPa matric potential) W_{WP} [mm], is calculated across the depth of the root-zone, z_R [mm]. The plant-available water PAW [mm] is then defined by a fraction *p* of TAW that a crop can extract from the root-zone without suffering water stress. Values of *p* are listed in Table 22 of Allen et al. (1998). The pattern of water and nutrient uptake from the root-zone soil is determined from the depth-wise pattern of root development (Green et al. 2002).

A.4. Modelling surface runoff

The surface runoff component of SPASMO is based on a daily rainfall total. The calculation uses the Soil Conservation Service (SCS) curve number approach (Williams 1991). The curve number approach was selected here because: (i) it is based on over 30 years of runoff studies on pasture, arable and forest sites in the USA, (ii) it is computationally simple and efficient, (iii) the required inputs are available, (iv) and the calculation relates runoff to soil type, land use and management practice.

Surface runoff is predicted from daily rainfall plus irrigation, using the SCS curve number equation:

$$\begin{aligned} Q &= \frac{(R-0.2S)^2}{R+0.8S}\,, && R > 0.2S \\ Q &= 0\,, \quad , && R \le 0.2S \end{aligned} \qquad \text{[Eq. A10]}$$

where Q [mm] is the daily runoff, R [mm] is the daily rainfall plus irrigation, and S [mm] is the retention parameter that reflects variations among soils, land use and management. The retention parameter, S, is related to the curve number, CN, using the SCS equation (Soil Conservation Service 1972)

$$S = 254\left(\frac{100}{CN} - 1\right) \qquad \text{[Eq. A11]}$$

where the constant, 254, gives S in millimetres. Moisture condition 2 (CN_2) or the average curve number, can be obtained easily for any area of land use type from the SCS Hydrology Handbook (Soil Conservation Service 1972). An example of CN numbers is given below for a range of pasture and drainage conditions.

Table A1. SCS curve number for a grazed pasture (Soil Conservation Service 1972).

SCS CN number		Drainage Condition			
		Excessive	Good	Fair	Poor
Pasture Condition	Good	39	61	74	80
	Average	49	69	79	84
	Poor	68	79	86	89

A pasture in good condition that is growing on a free draining soil will have a low CN value (39), while a pasture in poor condition and on a poorly drained soil will have a high CN value (89). A lower CN value implies a bigger retention parameter, S, and so a given soil/pasture combination will yield less runoff for the same daily rainfall total. The SCS runoff calculation also includes an additional adjustment to S, to express the effect of slope and soil water content (Williams 1991). In the calculations presented here, we have assumed the pasture slope is always less than 5% and have used a reference CN value for a pasture sward in average condition. The only other allowance that we have made, with respect to runoff, is to include any changes in S that are due to different soil water contents.

A.5. Nitrogen balance of the soil

The nitrogen component of SPASMO is based on a set of balance equations that account for nitrogen uptake by plants, exchange and transformation processes in the soil, losses of gaseous nitrogen to the atmosphere, additions of nitrogen in the effluent or fertilizer, and the leaching of nitrogen below the root zone. SPASMO considers both organic nitrogen (i.e. in the soil biomass) and the mineral nitrogen (i.e. urea, ammonium and nitrate). Dissolved urea and nitrate are considered to be mobile and to percolate freely through the profile, being carried along with the invading water. The movement of dissolved ammonium is retarded as it binds to the mineral clay particles of the soil. The soil can receive inputs of organic carbon and nitrogen from decaying plant residues, which is added to the litter layer of the topsoil, and inputs of ammonium and nitrate in the effluent applied to the soil surface. Details of the nitrogen component of SPASMO are published in Rosen et al. (2004).

A.6. Crop Growth

The uptake of soil nutrients (i.e. nitrogen and phosphorus) by pasture and trees is determined largely by the growth of the above- and below-ground DM, multiplied by their respective nitrogen concentrations. Daily biomass production is modelled using a potential production rate per unit ground area, G (kg m^{-2} d^{-1}) that is related, via a conversion efficiency, ε (kg MJ^{-1}), to the amount of solar radiant energy, Φ (MJ m^{-2} d^{-1}), intercepted by the leaves

$$G = \varepsilon \Phi \; f_T \;\, f_N \;\, f_W \qquad \text{[Eq. A12]}$$

Here f_T, f_N and f_W are response functions that range between zero and unity depending on temperature, plant nitrogen and soil water status respectively (Eckersten & Jansson 1991). The value of G depends on the daily sunshine and temperature, plus the leaf-area index of the crop, and is moderated by the soil's water and nitrogen status (King 1993; Thornley et al. 1995). Crop growth is maximised only if soil water and soil nutrients are non-limiting.

A simple allometric relationship is used to partition the daily biomass production into the growth of the foliage, stem material and roots. Plant biomass is expressed in terms of the balance between growth and senescence of the plant organs. For each plant organ we write out a simple mass balance equation that considers inputs of DM due to carbon allocation, losses of DM as the plants senescence, and the removal of DM as the plants are harvested. The total mass of foliage, F [kg m^{-2}] is calculated from

$$\frac{dF}{dt} = \alpha_F G - \gamma_F F - H_F \qquad \text{[Eq. A13]}$$

the total mass of stem material, S [kg m^{-2}] is calculated from

$$\frac{dS}{dt} = \alpha_S G - \gamma_S F - H_S \qquad \text{[Eq. A14]}$$

and the total mass of roots, R [kg m^{-2}], is calculated from

$$\frac{dR}{dt} = \alpha_R G - \gamma_R R \qquad \text{[Eq. A15]}$$

Here α_F is the fraction of biomass partitioned to the foliage, α_S is the biomass partitioned to the stem, and α_R (=1-α_F-α_S) is the fraction of biomass allocated to the roots, and γ is the corresponding senescence rate for these plant components. The variable *H* is used to represent the amount of DM that is removed during harvest. In the case of fruiting crops, additional terms are included in each balance equation to represent an amount of DM transferred to fruit production. These fruit-growth terms have been omitted from the balance equations because here we are considering only pasture and *Eucalyptus* trees.

Allocation of DM to the roots depends on the leaf nitrogen content $[N]_F$, having a minimum value [α_{R0}] at a maximum leaf concentration $[N]_{Fx}$, and increasing as N_L decreases (Eckersten & Jansson 1991)

$$\alpha_R = \alpha_{R0} + 1 - \left(1 - \left(\left([N]_{Fx} - [N]_F\right)/[N]_{Fx}\right)^2\right)^{0.5} \qquad \text{[Eq. A16]}$$

This formulation enables SPASMO to accommodate seasonal changes in DM allocation associated with a changing leaf nutrient status. For simplicity, any seasonal changes in senescence rates have been neglected in the model because we are concerned with the long-term consequences of DM allocation.

A.7. Nitrogen and Phosphorus Uptake

The model assumes plant growth will achieve a maximum potential only if water, nitrogen and phosphorus are non-limiting. The nitrogen demand for crop growth is set by the maximum nitrogen content of the root $[N]_{Rx}$, leaf $[N]_{Fx}$ and stem $[N]_{Sx}$ material. During active growth the plant tries to supply new DM material with nitrogen corresponding to these maximum concentrations. The potential uptake of nitrogen, U_N [kg ha^{-1} d^{-1}], is defined as

$$\begin{aligned} U_N = &\left(\alpha_F G[N]_{Fx} - \lambda_F \gamma_F F[N]_F\right) \\ &+\left(\alpha_S G[N]_{Sx} - \lambda_S \gamma_S S[N]_S\right) + \left(\alpha_R G[N]_{Rx} - \lambda_R \gamma_R R[N]_R\right) \end{aligned} \qquad \text{[Eq. A17]}$$

And this represents the new growth at the maximum N content minus an amount of nitrogen translocated from the senescing plant material. The potential (maximum) nitrogen uptake can only be met if sufficient nitrogen exists in the soil. Otherwise all [*N*]s will be reduced in low-nitrogen soils, and crop growth will be curtailed. The potential uptake of phosphorus uptake is modelled in the same way based on the maximum P content of the respective plant parts.

Daily uptake of nitrogen is assumed to be proportional to the local distribution of the fine roots, and the total amount of nitrate (NO_3^-) and ammonium (NH_4^+) in each soil layer (Johnsson et al. 1987). The potential uptake of nitrate is calculated as

$$U_{NO3-} = \min\left(\rho_R(z)\,\frac{NO_3^{\;-}}{NO_{3-} + NH_4^{\;+}}\,U_N, f_M NO_3^{\;-}\right) \qquad \text{[Eq. A18]}$$

based on the relative root fraction in the layer, $\rho_R(z)$, the proportion of total mineral nitrogen as nitrate, and the total growth requirement for nitrogen, U_N. However, the actual uptake of nitrate is limited to a fraction f_M [-] of the total nitrate available in each layer. Ammonium uptake is calculated in a similar way, being proportional to the relative amount of ammonium in solution.

Surface roots are the most active (Clothier & Green 1994) and they preferentially extract soil water and nutrients from the upper soil layers. However, as water and nitrogen stresses develop, the uptake activity typically switches to the deeper roots if more water and nutrients are available there. This feature of root action is modelled in the following way. Whenever the total nitrogen uptake from a given soil layer is less than the potential rate, then the model allows for a compensatory increase in uptake from remaining layers deeper in the root zone (Johnnson et al. 1987). This is achieved by adding a fraction c_{um} [-] of the deficit to the potential uptake from the next soil layer where more mineral nitrogen may be available.

Daily allocation of nitrogen to the new plant material is based on the idea that roots receive nitrogen first, until they reach their maximum concentrations, then nitrogen is allocated to the stem, and finally to the leaves. If soil nitrogen becomes limiting, a reduction factor f_N is used to reduce the total nitrogen uptake. This reduction function also effectively reduces the leaf nitrogen contents and alters the DM allocation pattern (Eckersten & Jansson 1991). A similar scheme is adopted for P uptake and allocation across the new plant material.

Pasture growth parameters in this study have been chosen to generate appropriate levels of DM production i.e. the model simulates between 10-15 Mg DM ha^{-1} yields from an irrigated pasture, and adds about 1000 kg DM for every 100 kg N ha^{-1} of nitrogen in the effluent.

A.8. Carbon and Nitrogen dynamics of the soil organic matter

The decomposition of soil biomass adds an amount of mineral nitrogen, in the form of ammonium, to the soil. This transformation process, known as mineralization, is modelled by dividing the soil's total organic matter into three pools – a fast cycling litter pool, an almost stable humus pool, and a manure pool (Johnsson et al. 1987). The relative amount of organic-N in these three pools changes daily to reflect inputs of fresh biomass, and manure, and the losses of soil biomass and plant residue as it decomposes. The nitrogen demand for this internal cycling of the soil's organic carbon and nitrogen is regulated by the C/N ratio of the soil biomass, r_O, which is one of the model inputs.

Decomposition of soil litter carbon (C_L) is modelled as a first-order process and is specified by a rate constant (K_L) that is influenced by temperature and soil moisture. The products of decomposition are CO_2, stabilized organic material (humus) and, conceptually, microbial biomass and metabolites. The relative amount of these products is determined by a synthesis efficiency constant (f_E) and a humification fraction (f_H). The following mass balance equations, which represent the inputs minus the outputs of soil-C and soil-N, are used to model the turnover of carbon and nitrogen in the litter pool

$$\frac{\partial C_L}{\partial t} = \left[(1-f_H)f_E - 1\right].K_L.C_L + F_{C,L}$$

$$\frac{\partial N_L}{\partial t} = \left[(1-f_H)f_E \frac{1}{r_O} - \frac{N_L}{C_L}\right].K_L.C_L + F_{N,L} \qquad \text{[Eq. A19]}$$

where F represents the amount of fresh organic matter that is added to the soil biomass. A similar set of equations describes the turnover of carbon and nitrogen in the manure pool (although this pool is not modelled here)

$$\frac{\partial C_M}{\partial t} = \left[(1-f_H)f_E - 1\right].K_M.C_M + F_{C,M}$$

$$\frac{\partial N_M}{\partial t} = \left[(1-f_H)f_E \frac{1}{r_O} - \frac{N_M}{C_M}\right].K_M.C_M + F_{N,M} \qquad \text{[Eq. A20]}$$

Lastly, the set of mass balance equations describing the turn-over of carbon and nitrogen in the humus pool are given by

$$\frac{\partial C_H}{\partial t} = f_E.f_H.K_L.C_L - K_H.C_H + F_{C,H}$$

$$\frac{\partial N_H}{\partial t} = \frac{f_E.f_H}{r_O} K_L.C_L - K_H.N_H + F_{N,H} \qquad \text{[Eq. A21]}$$

Decomposition of soil humus (C_H) follows first-order kinetics with a specific rate constant (K_H) that depends on temperature and soil moisture. The other terms in these mass balance equations have already been described above.

Soil carbon and nitrogen turn-over reactions result either in a net production (mineralization) or a net consumption (immobilization) of ammonium, depending on the C/N ratio of the soil biomass. From a consideration of mass balances, any increase in NH_4^+-N, due to mineralization, must equal the decrease in organic-N from the three organic matter pools. Thus, the following mass-balance equation is used to predict nitrogen mineralization

$$\frac{\partial NH_4^+}{\partial t} = \left[\frac{N_L}{C_L} - \frac{f_E}{r_O}\right] K_L.C_L + \left[\frac{N_M}{C_M} - \frac{f_E}{r_O}\right] K_M.C_M + K_H.N_H \quad \text{[Eq. A22]}$$

Net mineralization occurs whenever $\partial NH_4^+/\partial t > 0$, otherwise immobilization occurs. The calculations recognise that, if no ammonium is available for immobilization, then nitrate can be used according to the following equation

$$\frac{\partial NO_3^-}{\partial t} = -\frac{f_E}{r_O}\left(K_L.C_L + K_M.C_M\right) \quad \text{[Eq. A23]}$$

During all simulations reported here, literature values were adopted for most of the parameters: the rate constants were set equal to K_L=0.015 d^{-1}, K_M = 0.015 d^{-1} and K_H=0.00005 d^{-1}; constant values were used for the efficiency of carbon turn-over, f_E=0.5, the humification fraction, f_H=0.2, and the C/N ratio of the soil biomass, r_O=10.0, as suggested by Johnnson et al. (1987).

For the purpose of modelling, senescing plant material is added a single pool of organic P in the litter layer. The turnover of this organic phosphorus, to create mineral phosphorus (i.e. dissolved reactive phosphorus) is modelled simply by assuming decomposition is a first-order process specified by the rate constant K_L, and moderated by temperature and soil moisture functions.

A.11. Soil transformation processes for nitrogen

All N-transformation processes in the soil are assumed to be first-order with rate constants that are regulated by both temperature and moisture status of the soil. The effect of soil temperature is expressed using a Q_{10} relationship (Bunnell et al. 1977)

$$f_T(z) = Q_{10}^{\left[\frac{T(z)-T_B}{10}\right]} \quad \text{[Eq. A24]}$$

where $T(z)$ is the soil temperature for the layer, T_B is the base temperature at which f_T equals 1, and Q_{10} is the factor change in rate due to a 10-degree change in temperature. The soil moisture factor decreases, on either side of an optimum level, in drier soil or in excessively wet soil (Johnnson et al. 1987), i.e.

$$\begin{aligned}
f_W(z) &= f_S + (1-f_S)\left[\frac{\theta_S(z)-\theta(z)}{\theta_S(z)-\theta_H(z)}\right]^M && \theta_H(z) < \theta(z) < \theta_S(z), \\
f_W(z) &= 1 && \theta_L(z) < \theta(z) < \theta_H(z), \\
f_W(z) &= \left[\frac{\theta\ (z)-\theta_W(z)}{\theta_L(z)-\theta_W(z)}\right]^M && \theta_W(z) < \theta(z) < \theta_L(z), \qquad \text{[Eq. A25]}
\end{aligned}$$

where θ_S is the saturated water content, θ_H and θ_L are the high and low water contents, respectively, for which the soil moisture factor is optimal, and θ_W is the minimum water content for process activity. The factor f_S defines the relative effect of moisture when the soil is completely saturated, and M is an empirical constant.

The nitrogen model accounts for the internal cycling and transformation of three forms of mineral nitrogen (i.e. urea, ammonium and nitrate). The hydrolysis of urea (U, mg L^{-1}) to ammonium (NH_4^+, mg L^{-1}), is modelled as

$$\left.\frac{dU}{dt}\right|_{U\to NH4+} = -k_1\, f_T(z)\, f_M(z)\, NH_4^{\ +} \qquad \text{[Eq. A26]}$$

and this process is defined by a first-order rate constant (k_1). The transfer of ammonium to nitrate, (NO_3^-, mg L^{-1}), is modelled as

$$\left.\frac{dNH_4^{\ +}}{dt}\right|_{NH4+\to N03-} = -k_2\, f_T(z)\, f_M(z)\left[NH_4^{\ +} - \frac{NO_3 -}{n_q}\right] \qquad \text{[Eq. A27]}$$

and depends on the potential rate constant (k_2) which is reduced as the nitrate-ammonium ratio (n_q) of the soil is approached. If $NH_4^+ < NO_3^- / n_q$ then no transfer of ammonium to nitrate takes place.

Denitrification is the transfer of nitrate to gaseous nitrogen (N_2 and N_2O) products. This is an anaerobic process and consequently is highly dependent on soil aeration. Soil water content is used as an indirect expression of the oxygen status of the soil. The influence on the denitrification rate is expressed as a power function

$$f_D(z) = \left[\frac{\theta(z) - \theta_D(z)}{\theta_S(z) - \theta_D(z)}\right]^d \qquad \text{[Eq. A28]}$$

that increases from a threshold point (θ_D), is maximum at saturation (θ_S), and d is an empirical constant. No denitrification occurs below the threshold point. The denitrification rate for each layer is modelled as

$$\left.\frac{dNO_3^{\ -}}{dt}\right|_{NO3-\to gas} = -k_3(z)\, f_T(z)\, f_D(z)\left[\frac{NO_3^{\ -}}{NO_3^{\ -} + c_S}\right] \qquad \text{[Eq. A29]}$$

and depends on a potential denitrification rate (k_3), the soil aeration status (f_D), and the same temperature factor (f_T) used for the other biologically-controlled processes. The rate constant k_3 is assumed to be a linear function of soil organic-carbon (Smith & Arah 1990). The factor c_S is the nitrate concentration where the denitrification rate is 50% of the maximum, all other factors are optimum.

The ammonia volatilization model incorporates the effect of soil and effluent pH, soil and air temperature, wind speed, and soil water content (Smith et al 1996). The following mechanistic equation of Wu et al. (2003) is used to prescribe the soil-surface volatilization rate, J_V [kg m^{-2} s^{-1}], as

$$J_V = \left(\frac{\theta_0}{\theta_S}\right) h_M \left.\left(\frac{K_A K_H}{10^{-pH}}\, NH_4^{\ +}\right)\right|_{z=0} \qquad \text{[Eq. A30]}$$

where h_M is the average mass transfer coefficient, K_A is the equilibrium constant relating the concentrations of ammonium ion and dissolved ammonia in soil solution, K_H is Henry's constant for the dissolution of gas-phase and liquid-phase ammonia in soil solution. The formulation for these three factors is presented in Wu et al. (2003).

A.12. Mass-balance equations for mineral nitrogen and phosphorus and microbes

The nitrogen transport model allows for an input of mineral nitrogen in the form of urea, ammonium or nitrate. The fate of surface-applied urea is determined by two competing processes:

- Losses due to hydrolysis of urea to ammonia
- Losses due to the drainage of urea through the soil profile.
-

We have assumed that all the urea enters the soil, and that any surface runoff of urea is negligible. The total mass of urea M_U [mg m^{-2}], in each soil slab of thickness z_R [mm] is found by solving the following mass balance equation

$$\frac{dM_U}{dt} = z_R \frac{d\theta R_U U}{dt} = X_{U,i} - \left(k_1 z_R \theta U + J_W U\right) \qquad \text{[Eq. A31]}$$

where U [mg L^{-1}] is the concentration of urea in soil solution, θ [m^3 m^{-3}] is the soil's volumetric water content, $X_{U,i}$ [mg m^{-2}] is the mass of urea added to the i-th segment (=0 if i >1), k_1 [d^{-1}] is the rate-constant describing the hydrolysis of urea to ammonium, $J_W U$ [mm d^{-1}] represents the percolation of dissolved urea through the soil. Urea is rapidly hydrolysed to ammonium, in a matter of a few days. The fate of ammonium-nitrogen is determined by six competing processes:

- Inputs from the mineralization of the soil biomass
- Retardation due to the adsorption of ammonium to the soil particles
- Losses due to the nitrification of ammonium into nitrate
- Losses due to the volatilization of ammonia gas
- Losses due to the drainage of ammonium through the soil slab
- Losses due to plant uptake.

The total mass of ammonium, M_A [mg m^{-2}], in each soil slab is found by solving the following mass balance equation:

$$\begin{aligned}\frac{dM_A}{dt} = z_R \frac{d\theta R_A A}{dt} &= \left(X_{A,i} + S_M + k_1 z_R \theta\, U\right) \\ &- \left(k_2 z_R \theta\left(A - N/n_q\right) + J_V + P_A + J_W A\right)\end{aligned} \qquad \text{[Eq. A32]}$$

where A [mg L^{-1}] is the concentration of ammonium in soil solution, $X_{A,i}$ [mg m^{-2}] is the total mass of ammonium added to the i-th layer (=0 if i >1), S_M [mg m^{-2}] is rate of mineralization, P_A [mg m^{-2} d^{-1}] is the rate of plant uptake, k_2 [d^{-1}] is a rate constant to describe the nitrification of ammonium to nitrate, and $J_W A$ [mg m^{-2} d^{-1}] represents the percolation of dissolved ammonium through the soil slab. Here J_V represents the volatilisation of ammonia to the atmosphere. For simplicity, we have calculated J_V only for the top 10 cm of soil and set it equal to zero elsewhere. R_A = (1 +ρK_D/θ) is the retardation factor for ammonium, ρ [kg L^{-1} soil] is the soil's dry bulk density, and K_D [L kg^{-1}] is the distribution coefficient that determines how much ammonium gets adsorbed to the cation-exchange sites of the soil.

The fate of any nitrate in the soil water is determined by the following six processes:

- Inputs of nitrate from fertilizer application

- Inputs from the nitrification of ammonium
- Retardation due to the adsorption of nitrate (= 0 in most mineral soils)
- Losses from denitrification
- Losses due to plant uptake
- Losses due to the drainage of nitrogen beyond the root zone.

The total mass of nitrate-nitrogen, M_N [mg m^{-2}], in each soil slab is found by solving the following mass balance equation

$$\frac{dM_N}{dt} = z_R \frac{d\theta R_N N}{dt} = \left(X_{N,i} + k_2 z_R \theta A\right) - \left(k_3 z_R \theta \left[\frac{N}{N + c_S}\right] + P_N + J_W N\right) \qquad \text{[Eq. A33]}$$

where N [mg L^{-1}] is the concentration of nitrate in soil solution, $X_{N,i}$ [mg m^{-2}] is the total mass of nitrate-nitrogen added to the *i*-th layer (=0 if i >1), k_3 [d^{-1}] is a rate constant to describe denitrification losses, P_N [mg m^{-2} d^{-1}] is the rate of plant uptake, and $J_W N$ [mg m^{-2} d^{-1}] represents the drainage of nitrate through the soil slab. We consider denitrification to be a microbial process that is rate-limited by the amount of soil organic carbon (the energy source) and mineral nitrogen (the nutrient source).

The total mass of mineral phosphorus, M_P [mg m^{-2}], in each soil slab is found by solving the following mass balance equation

$$\frac{dM_P}{dt} = z_R \frac{d\theta R_P P}{dt} = \left(X_{P,i} + S_P - P_P + J_W P\right) \qquad \text{[Eq. A34]}$$

where P [mg L^{-1}] is the concentration of dissolved reactive phosphorus in soil solution, $X_{P,i}$ [mg m^{-2} d^{-1}] is the total mass of phosphorus added to the *i*-th layer (=0 if i >1), S_P [mg m^{-2} d^{-1}] is the rate of mineralization of organic phosphorus, P_P [mg m^{-2} d^{-1}] is the rate of plant uptake, and $J_W P$ [mg m^{-2} d^{-1}] represents the drainage of dissolved phosphorus through the soil slab. The adsorption of phosphorus is modelled using a Langmuir isotherm (Eq. A5), and so the retardation for phosphorus, R_P, is calculated as

$$R_P = \left[1 + \left(\frac{\rho}{\theta}\right)\left(\frac{Qb}{1 + bP}\right)\right] \qquad \text{[Eq. A35]}$$

where Q is the maximum total mass of phosphorus at saturation per unit mass of dry soil [µg g^{-1}], and b is an empirical constant, with units of inverse of solution concentration [L mg^{-1}].

Bacterial transport is calculated using the same convection-dispersion type equation for water and solute transport, with additional terms used to represent the kinetic sorption of bacteria to the soil's mineral particles as well as the subsequent detachment and transfer of bacteria between the aqueous and solid phases (Schijven and Hassanizadeh 2000). The mass balance equation for water-borne bacteria (considering only those bacteria applied in the effluent) is given by the following equation

$$\frac{d\theta B}{dt} = \left(X_{B,i} - k_a \theta \psi B + k_d \rho S - \mu_W \theta B - \mu_S \rho S - J_W B\right) \qquad \text{[Eq. A36]}$$

where B represents the bacteria concentration in the liquid phase [cfu L^{-1}], S_B represents the bacteria concentration in the solid (sorbed) phase [cfu g^{-1}], $X_{B,i}$ is the total mass of bacteria added to the i-th layer (=0 if i >1) [cfu m^{-2} d^{-1}], the k_a term represents attachment of bacteria to the soil particles, and the k_d term represents detachment of bacteria from the soil particles and $J_W B$ [mg m^{-2} d^{-1}] represents the drainage of bacteria through the soil slab. The inactivation (die-off) of bacteria is described using a simple first-order decay model, where μ is the mortality rate [d^{-1}] and the subscripts 'w' and 's' refer to the liquid and solid phases, respectively. The overall mortality rate for *E. coli* bacteria in soil has been reported to be between 0.09 and 0.17 d^{-1} in two contrasting silt loams (Mubiru et al. 2000). Sukias & Nguyen (2003) report the rate constant for bacterial die-off in a Te Kowai silt loam, Hamilton, is about 0.056 d^{-1}. This represents a 'half life' of between about 1.8 and 3.3 days.

Calculation procedure

The model is run using a daily time step, to track the fate of nutrients and contaminants in effluent-applied land. The model considers the 11 irrigation areas separately, adding different amount of effluent to each site depending on pond disposal requirements and set irrigation rules. The calculations are made in the following sequence:

- Subtract evaporation, transpiration and plant uptake of nutrients from each soil segment
- Add and subtract the nitrogen, phosphorus and bacteria involved in the various transformation processes
- Partition each contaminant between solution and sorbed fractions, assuming complete equilibrium between the mobile and immobile phases
- If there is rain or irrigation, then perform the leaching process
- Redistribute water and contaminants vertically, according to water potential and solution concentration
- Repeat the contaminant partitioning.

APPENDIX B: MODEL INPUTS

The following data sources were used to obtain time series inputs for the model

- Climate data were obtained from NIWA's CLIFLO (www://cliflo.niwa.co.nz) database using records from Te Ore Ore (1997-2006) weather station. The data included daily values of incoming global radiation, air temperature and relative humidity, maximum and minimum air temperatures, wind speed and rainfall. Occasional missing values were replaced with corresponding data from the NIWA climate station at Masterton.
- River flow data (daily and hourly mean flows, L s^{-1}) were obtained from flow gauge records at Wardell's Bridge (data supplied by Beca). Occasional missing values were replaced with estimates derived from a weighted sum of the daily rainfall recorded over the previous 3 weeks. Suitable weighting factors were found using the SOLVER routine in Microsoft® Excel®.
- Daily effluent volumes and composition, including the solution concentration of nitrogen (as nitrate and ammonium), dissolved reactive phosphorus, and *E. coli* were supplied by Beca. For the purpose of calculation, the following concentrations were assumed in the winter time; *E. coli* = 1000 cfu per 100 ml, total mineral nitrogen = 11.5 mg L^{-1} (half as ammonium and half as nitrate), dissolved reactive P = 2.5 mg L^{-1}. We assumed the same concentrations in the summer except *E. coli* = 200 cfu per 100 ml.

The soil's hydraulic (i.e. water retention and drainage capacity) and physical (i.e. bulk density, texture, organic C & N) properties, and the transport properties for P were determined using soil samples taken from a large number of auger holes and soil pits across the site (Figure B1). The corresponding soil property data are described in Section 3.

The modelling framework for the SPASMO is presented in Appendix A as a series of equations describing the transport, partitioning, and fate of water and contaminants (N, P & bacteria), as well as the growth and uptake of nutrients (N & P) by plants receiving effluent wastewater. Table B1 below summarises the model's input variables and provides either a parameter value or a reference to each of the equations used in the model.

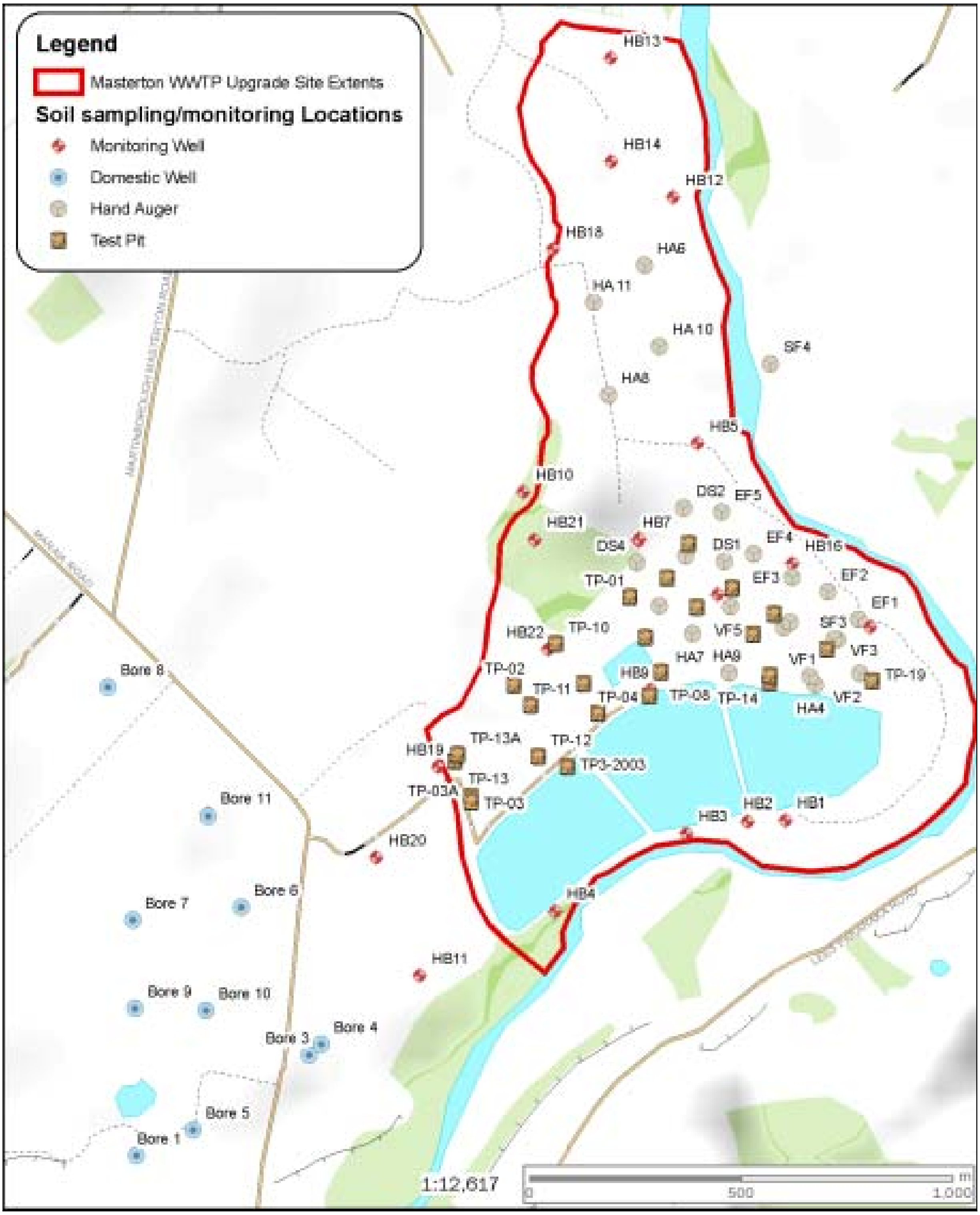

Figure B1. Locations of wells, test pits and bore holes at and in the vicinity of the Masterton Wastewater Treatment Plant. Soil data from these sample locations were used to derive a map showing zones of similar soil properties (Hugh Wilde, Landcare Research Ltd, Palmerston North).

Table B1. Adopted values for each parameter used in SPASMO model (see text for details).

Parameter	Model component	Value or Equation	Unit
Climate			
c_P	Specific heat capacity of air	1010	J kg^{-1} m^{-3}
e_A	Vapour pressure of ambient air	A8	Pa
e_S	Saturated vapour pressure at ambient temperature	A8	Pa
G	Ground heat flux	A8	W m^{-2} d^{-1}
R_N	Net radiation	A8	W m^{-2} d^{-1}
RH	Relative humidity	A8	%
R_F	Daily rainfall	A10	mm
T_A	Air temperature	-	^oC
W_S	Average daily wind speed	-	m s^{-1}
ρA	Air density	A8	kg m^{3}
Pond			
L_R	Leakage rate	490	m^3 d^{-1}
Q_M	Maximum effluent discharge	104,000	m^3 d^{-1}
S	Storage volume	-	m^3
S_M	Maximum storage volume	363,000	m^3
V_P	Pond volume	-	m^3
W	Base width	502	m
Z_P	Pond working volume	-	m^3
Δ	Site slope	1:2.5	-
Soil			
C_I	Solution concentration in the mobile domain	A2	mg L^{-1}
C_M	Solution concentration in the immobile domain	A2	mg L^{-1}
M_C	Total amount of solute	A2	mg m^{-2}
PAW	Plant available water	A9	mm
S	Sorbed solute concentration	A3	mg kg^{-1}
T_S	Soil temperature	A24	^oC
TAW	Total available water, WFC-WWP	A9	mm
W_{FC}	Water content at field capacity (-10 kPa matric pot.)	A9	mm
W_{SP}	Water content at stress point (~-100 kPa matric pot.)	A9	mm
W_{WP}	Water content at wilting point (-1500 kPa matric pot.)	A9	mm
W_I	Water content in immobile domain	A9	mm
W_M	Water content in mobile domain (> -100 kPa matric pot.)	A9	mm
W_T	Total water content	A1	mm
d_C	Mean grain size diameter of porous material	A7	mm
θ	Volumetric water content	A6	L L^{-1}
θ_S	Saturated vol. water content	A25	L L^{-1}
θ_N	Residual water content	A25	L L^{-1}
ρ	Bulk density	A4	kg m^{-3}
Solute transport and transformation			
b	Half saturation point for phosphorus	A5	mg L^{-1}
c_S	Half saturation point for denitrification	A29	mg L^{-1}
d	Empirical factor for denitrification	A28	-
h_M	Average mass transfer coefficient for volat. losses	A30	-
$f_D(\theta)$	θ-response function for denitrification	A28	-
f_S	Relative effect of θ on abiotic processes	A25	L L^{-1}
J_V	Volatilization losses of NH_3 gas	A30	mg m^{-2}
J_W	Water flux density	A31	mm d^{-1}
J_S	Solute flux density	A32	mg m^{-2} d^{-1}
K_D	Solute distribution coefficient	A3	L kg^{-1}

k_1	Rate constant for hydrolysis of Urea → NH_4^+	A31	d^{-1}
k_2	Rate constant for nitrification of NH_4^+ → NO_3^-	A32	d^{-1}
k_3	Rate constant for denitrification NO_3^- → N_20	A33	d^{-1}
K_A	Equilibrium constant NH_4^+(aq) → NH_3(gas)	A30	-
K_H	Henry's constant	A30	-
M	Empirical factor for abiotic functions	A25	-
M_A	Total mass of ammonium in the soil	A32	mg m^{-2}
M_N	Total mass of nitrate in the soil	A33	mg m^{-2}
M_P	Total mass of phosphorus in the soil	A34	mg m^{-2}
M_U	Total mass of urea in the soil	A31	mg m^{-2}
P_A	Plant uptake of ammonium	A32	L m^{-2}
P_N	Plant uptake of nitrate	A33	L m^{-2}
P_P	Plant uptake of phosphorus	A34	L m^{-2}
P_U	Plant uptake of urea	A31	L m^{-2}
R_A	Retardation factor for ammonium	A32	L m^{-2}
R_N	Retardation factor for nitrate	A33	L m^{-2}
R_P	Retardation factor for phosphorus	A34	L m^{-2}
R_U	Retardation factor for urea	A31	L m^{-2}
S_P	1st order rate constant for mineralization of org. P	0.01	d^{-1}
U	Solution concentration of urea	A31	mg L^{-1}
X_A	Mass of ammonium added to the soil	A32	mg m^{-2}
X_N	Mass of nitrate added to the soil	A33	mg m^{-2}
X_P	Mass of phosphorus added to the soil	A34	mg m^{-2}
X_U	Mass of urea added to the soil	A31	mg m^{-2}
n_q	Ratio of soil nitrate: ammonium	A27	mg mg^{-1}
Q_P	Soil P concentration at saturation	A5	mg kg^{-1}
θ_D	Threshold water content for denitrification	A25	L L^{-1}
θ_H	High water content limit for abiotic response	A25	L L^{-1}
θ_L	Low water content limit for abiotic response	A25	L L^{-1}
Bacterial transport			
B	Bacteria concentration in the liquid phase	A34	cfu/L
d_p	Bacteria particle size	B2	mm
k_a	1st order attachment coefficient	A6	d^{-1}
k_d	1st order detachment coefficient	A6	d^{-1}
R_B	Retardation factor for bacteria	A35	L m^{-2}
X_B	Mass of bacteria added to the soil	A34	cfu m^{-2} d^{-1}
α	Collision efficiency	A7	-
η	Collection efficiency	A7	-
μ_S	Mortality rate (die-off) of the solid phase	0.1	d^{-1}
μ_W	Mortality rate (die-off) of the liquid phase	0.1	d^{-1}
υ	Pore water velocity	A7	mm d^{-1}
ψ	Colloidal retention function	A7	-
Crop water use			
ET_C	Crop evapotranspiration	A9	mm d^{-1}
ETo	Reference evapotranspiration	A8	mm d^{-1}
K_C	Crop factor	A9	-
K_R	Water stress factor	A9	-
LAI	Leaf area index	A12	m^2 m^{-2}
p	Drought tolerance	A9	-
r_A	Aerodynamic resistance	A8	s m^{-1}
r_S	Surface resistance	A8	s m^{-1}
s	Slope of the vapour pressure and temperature curve	A8	Pa K^{-1}
Z_R	Root depth	A9	mm
U_2	Wind speed at 2 m	A8	m s^{-1}
λ	Latent heat of vaporization	2.45	MJ kg^{-1}
γ	Psychometric constant	66.1	Pa K^{-1}

Crop growth			
f_T	Temperature response function	A24	-
f_N	Nitrogen response function	A12	-
f_W	Water response function	A25	-
F	Foliage dry matter content	A13	kg m^{-2}
G	Potential growth rate	A12	kg m^{-2} d^{-1}
H	Amount of dry matter (DM) removed during harvest	A13	^{K}g m^{-2}
R	Root dry matter content	A15	kg m^{-2}
S	Stem dry matter content	A14	kg m^{-2}
U_N	Potential uptake of nitrogen	A18	mg m^{-2} d^{-1}
P_A	Plant uptake of ammonium	A32	mg m^{-2} d^{-1}
P_N	Plant uptake of nitrate	A33	mg m^{-2} d^{-1}
P_P	Plant uptake of phosphorus	A34	mg m^{-2} d^{-1}
α_F	DM allocation to foliage	A13	-
α_S	DM allocation to stem	A14	-
α_R	DM allocation to root	A15	-
γ_F	Senescence rate of foliage	0.01	d^{-1}
γ_S	Senescence rate of stem	0.00001	d^{-1}
γ_R	Senescence rate of roots	0.0025	d^{-1}
ε	Conversion efficiency	A12	kg m^{-2} MJ^{-1}
Φ	Intercepted solar radiation	A12	MJ m^{-2} d^{-1}
ρ_R	Relative root fraction	A18	-
Runoff variables			
Q_R	Runoff volume	A10	mm d^{-1}
S	Surface water retention parameter	A10	mm
CN	Soil Conservation Service curve number	A11	-
CN_2	Average curve number	A11	-
Soil organic matter			
C_H	Soil carbon content as humus	A21	mg m^{-2}
C_L	Soil carbon content as litter (labile carbon)	A19	mg m^{-2}
C_M	Soil carbon content as manure	A20	mg m^{-2}
F_C	Fresh C added to the soil biomass	A19	mg m^{-2}
F_N	Fresh N added to the soil biomass	A19	mg m^{-2}
K_H	1^{st} order rate constant for decomposition of humus	A21	d^{-1}
K_L	1^{st} order rate constant for decomposition of litter	A19	d^{-1}
K_M	1^{st} order rate constant for decomposition of manure	A20	d^{-1}
N_L	Soil organic N as litter	A19	mg m^{-2}
N_H	Soil organic N as humus	A21	mg m^{-2}
N_M	Soil organic N as manure	A20	mg m^{-2}
Q_{10}	Temperature response factor	2.0	-
T_B	Base temperature for abiotic response function	15	C
r_o	C:N ratio of soil biomass	10	mg C mg^{-1}N
f_e	Microbial efficiency	0.2	-
f_h	Humification fraction	0.4	-

Phosphorus sorption isotherm

Model parameters describing phosphorus partitioning were determined using isotherm experiments performed on soil samples that were previously washed and dried at room temperature. The work was carried out by Landcare Research and the raw data were provided to HortResearch (Hugh Wilde, pers. comm.) in order to calculate the appropriate transport properties for P. The soil analysis was as follows:

Approximately 3 g dry soil was placed in 100-ml polythene bottles. Aliquots (30 ml) of a distilled water solution of 0.1 M KNO_3 solution spiked with KH_2PO_4 to give one of five levels of phosphorus (0, 1.0, 2.5, 10, 50, 100 mg L^{-1}) were then added. The bottles were sealed and continuously agitated in a rotating wheel at laboratory temperature for 16 hours. After settling for 1 hour, an aliquot of the supernatant was filtered through Whatman GF/C filters and, after adequate dilution, analysed for phosphorus by using the modified EPA method (USEPA 1982) with a detection limit of 0.0001 µg P ml^{-1}. P removal and equilibrium P-concentration data from the isotherm studies are presented below.

Table B2. Equilibrium P-concentration in soils from the Homebush, Masterton waste treatment plant.

Location/horizon/depth	Sample No.	Final conc with 1 mg/L added	Final conc with 2.5 mg/L added	Final conc with 10 mg/L added	Final conc with 50 mg/L added	Final conc with 100 mg/L added
Bore Ap 0-10	M4/0153	0.257	0.374	1.52	22.95	63.17
MS Northeast Bw 33-36	M4/3105	0.060	0.088	0.78	21.90	65.75
MS Northeast Bw2 58-61	M4/3106	0.060	0.083	1.06	24.19	71.91
Bush Ap 0-10	M4/0147	0.156	0.219	0.93	17.23	51.15
MS Bush1 Bw 50-53	M4/3107	0.062	0.069	0.43	16.60	57.12
MS Bush1 2Bg 91-94	M4/3108	0.056	0.067	0.22	14.69	55.01
Enclosure Ap 0-10	M4/0143	0.100	0.131	0.52	14.11	47.07
MS Enclosure Bw1 48-51	M4/3109	0.058	0.078	0.76	20.82	67.32
MS Enclosure Bw2 77-80	M4/3110	0.059	0.083	0.94	23.78	70.57
MS Pumphouse Ap 11-14	M4/3111	0.071	0.090	0.52	16.76	55.40
MS Pumphouse Bw 40-43	M4/3112	0.066	0.115	1.93	29.49	74.98
MS Pumphouse 2Bg 61-64	M4/3113	0.055	0.070	0.33	16.24	55.91

Data from the isotherm studies (Tables B2 and B3) were fitted to the Langmuir adsorption-isotherm equation of the form

$$\frac{q}{Q} = \frac{bC}{1+bC} \qquad \text{[Eq. B1]}$$

where C is the equilibrium adsorbate concentration [mg L^{-1}], q is the mass of adsorbate per mass of adsorbent at equilibrium [µg g^{-1}], Q is the maximum mass adsorbed at saturation conditions per mass unit of adsorbent [µg g^{-1}], and b is an empirical constant with units of inverse of concentration [L mg^{-1}]. Table B3 presents parameter values derived from soil samples collected at four locations on the Homebush site. The samples span a range of soil textures and soil depths.

Table B3. Equilibrium P-removal in soils from the Homebush, Masterton waste treatment plant.

Location/horizon/depth	Sample No.	P sorbed with 10 mg/kg added	P sorbed with 25 mg/kg added	P sorbed with 100 mg/kg added	P sorbed with 500 mg/kg added	P sorbed with 1000 mg/kg added
		(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
Bore 0-10	M4/0153	7.431	21.261	84.77	270.46	368.33
MS Northeast Bw 33-36	M4/3105	9.404	24.121	92.24	280.95	342.45
MS Northeast Bw2 58-61	M4/3106	9.400	24.172	89.37	258.11	280.89
Bush 0-10	M4/0147	8.436	22.812	90.69	327.70	488.47
MS Bush1 Bw 50-53	M4/3107	9.383	24.315	95.66	334.01	428.81
MS Bush1 2Bg 91-94	M4/3108	9.437	24.335	97.83	353.14	449.89
Enclosure 0-10	M4/0143	9.002	23.691	94.76	358.91	529.28
MS Enclosure Bw1 48-51	M4/3109	9.421	24.223	92.38	291.84	326.78
MS Enclosure Bw2 77-80	M4/3110	9.411	24.173	90.62	262.23	294.35
MS Pumphouse Ap 11-14	M4/3111	9.287	24.101	94.81	332.36	445.96
MS Pumphouse Bw 40-43	M4/3112	9.340	23.854	80.68	205.13	250.20
MS Pumphouse 2Bg 61-64	M4/3113	9.450	24.303	96.73	337.61	440.88

Parameters determining P-retention at other locations on the Homebush site were estimated from local soil data in the following way. Firstly, there is very good spatial information on the depth-wise profile of soil texture across the site (Table B1). There is also a very good soil database relating soil texture to hydraulic properties such as the water holding capacity (Table B2). Using this data we then sought a relationship between soil texture and the P transport properties (i.e. Q and B of Table B3). A simple quadratic equation was first fitted to data describing the relationship between the soil's field capacity, as estimated from texture, and %P retention as measured from soil samples (Figure B@). In general, the free-draining sandy soils hold less water and have a lower value for %P retention than the poorly-drained silty clay loams. Figure B1 enables us to estimate a value for phosphorus retention across the site, where we have good soil textural information but where %P retention data are lacking. About 75% of the scatter in the data is explained by the fitted line.

Table B4. Phosphorus retention in soils from the Homebush site (adapted from the data of Wilde and Dando (2005)). Two modelling parameters describe the soil's capacity to retain phosphorus. *Q* represents the maximum soil concentration at phosphorus saturation [mg kg^{-1}], and *B* represents the solution concentration [mg L^{-1}] where the soil is at 'half-saturation' with respect to phosphorus.

Location	Horizon	Depth [cm]	Texture	P retention [%]	Q [mg kg^{-1}]	B [mg L^{-1}]
Northeast Bore	Ap	0-10	silt loam	13	398.4	8.2
	Bw	33-36	silty clay loam	15	333.5	2.2
	Bw2	58-61	loamy silt	11	286.4	2.3
Bush	Ap	0-10	silt loam	19	587.4	11.8
	Bw	50-53	clay loam	19	409.1	1.7
	2Bg	91-94	silt loam	19	419.0	0.9
Enclosure	Ap	0-10	silty clay loam	20	615.1	8.8
	Bw1	48-51	silty clay loam	13	328.5	2.0
	Bw2	91-94	sandy loam	11	295.0	2.1
Pumphouse	Ap	11-14	silty clay loam	16	431.6	2.6
	Bw	40-43	loamy fine sand	8	251.5	4.3
	2Bg	61-64	silty clay loam	19	411.7	1.3

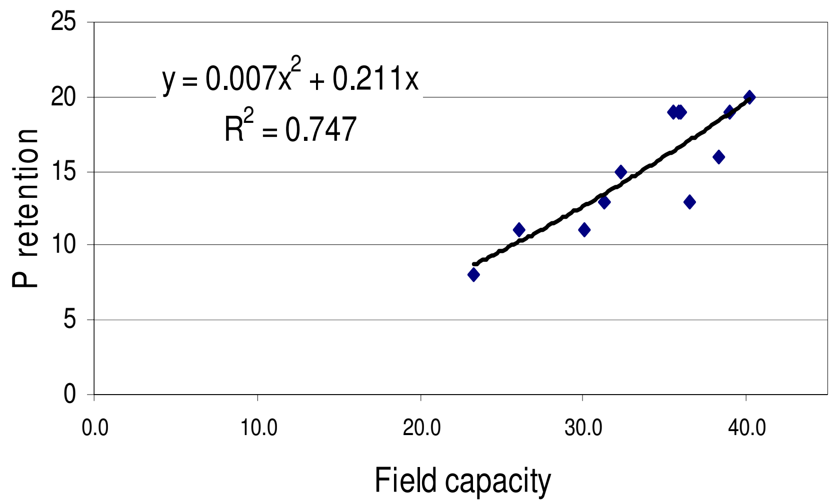

Figure B2. The relationship between the Homebush, Masterton soil's field capacity, as estimated from texture, and the %P retention was fitted using a quadratic equation.

Next we sought a relationship between %P retention and the maximum soil concentration at phosphorus saturation, as represented by the *Q*-value [mg kg^{-1}] of Equation B1. In general, *Q* increases with increasing %P, and a simple linear equation provides a good fit to the data (Figure B3). The maximum soil concentration ranged between about 250-600 mg kg^{-1}, with lowest values in the sands and highest values on the silty clay loams. Finally, a scatter plot revealed a tendency for the remaining isotherm parameter, *b* [L mg^{-1}], to increase with increasing soil depth (Figure B4). About 81% of the scatter in the data is explained by the fitted line.

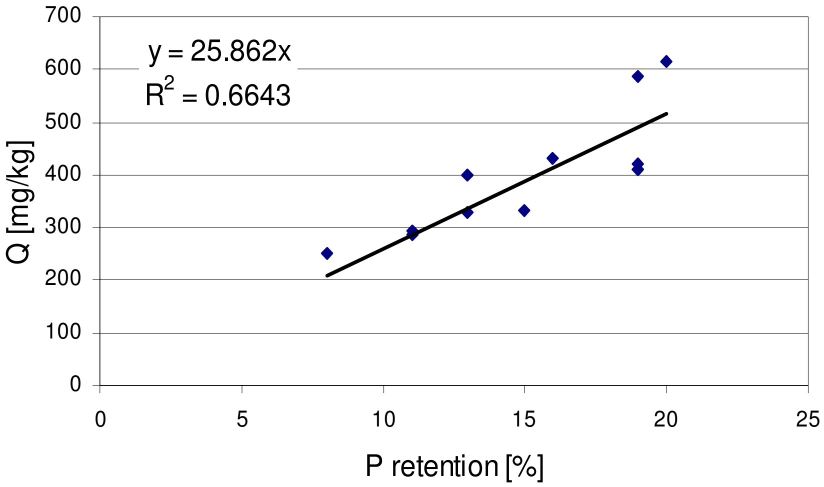

Figure B3. The relationship between the %P retention of the Homebush, Masterton soil and the maximum soil concentration at phosphorus saturation, Q-value [mg kg^{-1}], was fitted using a linear least-square regression (line).

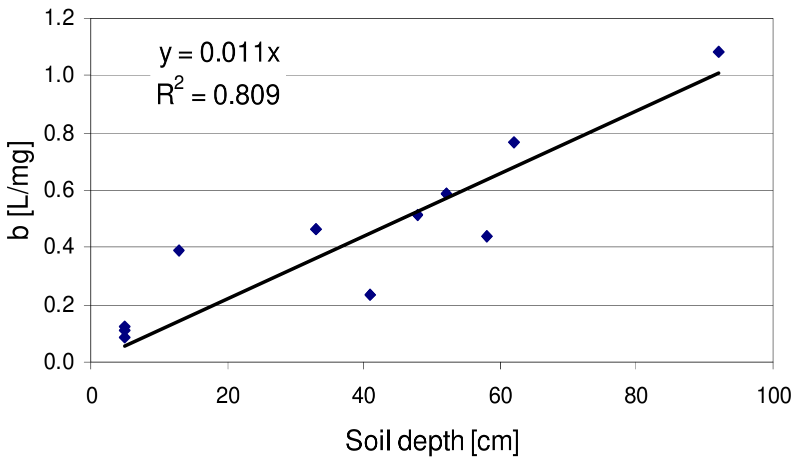

Figure B4. The relationship between soil depth at the Homebush, Masterton site and the isotherm parameter b (L mg^{-1}) was fitted using a linear least-squares regression (line).

Figures B2-B4 summarise the simple pedotransfer approach used here to determine phosphorus transport behaviour in the Homebush soils. From soil texture and depth alone we can estimate the P transport properties with reasonable accuracy. For example, in Figure B5 data from the Homebush site (BW horizon of clay loam) are compared against the Langmuir isotherm derived from soil texture and soil depth. The modelling efficiency of the estimation procedure, M_E as defined by $1-\Sigma[x-x_P]^2/\sigma^2$ [x=data, x_P = prediction, and σ = standard deviation] was 98%. For most of the isotherms presented in Table B4, M_E varied between 93 and 99%. The exception was the Bw1 horizon (silty clay loam) at the Enclosure site, where M_E=0.79%.

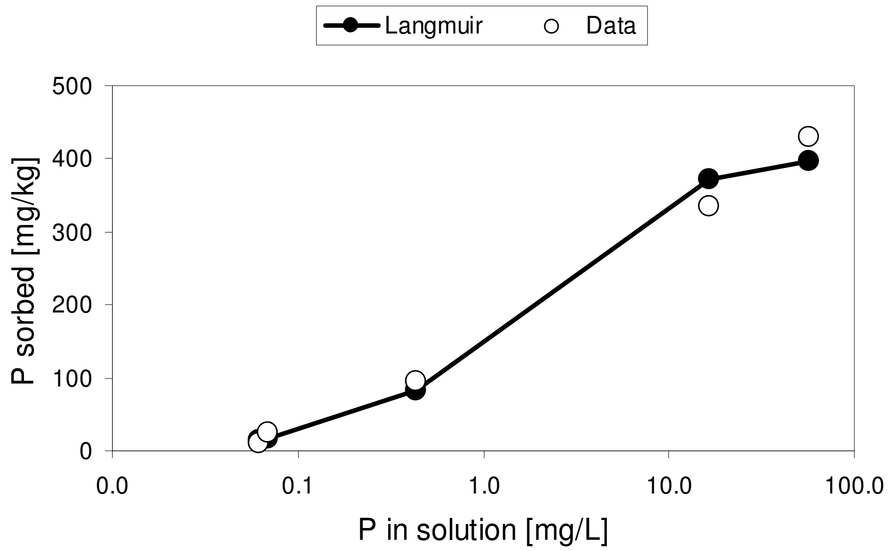

Figure B5. Langmuir isotherm for P retention in soil from the Bw horizon (50-53 cm deep) at the Homebush, Masterton site. The modelling efficiency was 98% (see text for details).

Colloidal filtering of the soil particles

As described in Appendix A, the water-borne transport of bacteria through soil was modelled as a colloidal-filtration process that depends on soil texture and an inactivation (die-off) rate that is determined by a characteristic half-life (residence time). The mortality rate (die-off) is modified by the temperature (Eq. A24) and water content of the soil (Eq. 25). The main soil parameter used to describe the filtering of bacteria is the mean particle size, and this was determined from soil texture in the following way. Firstly, the USDA has developed a unified textural classification system based on the soil's percentage of clay, silt and sand. The common approach of mechanical analysis (sieving) separates soil components into a range of particle sizes defined as

- Sand: $0.050 < d_p < 2.000$ mm
- Silt: $0.002 < d_p < 0.050$ mm
- Clay: $0.000 < d_p < 0.002$ mm.

The geometric particle size diameter is then calculated as $d_g = \exp(a)$, where the factor $a = 0.01\sum_{i=1}^{n} f_i \ln(M_i)$. Each f_i represents the percent of clay, silt and sand, and M_i is the corresponding arithmetic mean of the particle diameter. The mean particle size for a range of soil textures at the Homebush site is presented in Table B5. The attachment coefficient for bacteria is related to the mean grain diameter (i.e. particle size), as described below.

Table B5. The relationship between mean particle size (mm) and soil texture, as determined from the volume fractions (VF) of clay, silt and sand (from Shirazi and Boersma 1984).

Texture	VF clay	VF silt	VF sand	mean size (mm)
clay	50	20	30	0.015
silty clay loam	34	50	16	0.015
loamy silt	20	60	20	0.028
silt loam	20	60	20	0.028
clay loam	34	33	33	0.029
sandy clay loam	28	12	60	0.095
silty sand	20	25	55	0.102
sandy loam	10	25	65	0.205
loamy sand	6	12	82	0.435
sand	5	3	92	0.649

The attachment of bacteria in flowing water to the surfaces of solid particles in a porous medium involves two processes: mass transport to the surface, and surface interactions. These processes are described using colloid filtration theory, by expressing the attachment rate in terms of a collision efficiency η and a sticking efficiency α. According to this theory, a suspended particle may come into contact with a particle of the solid medium, the collector, either by interception, sedimentation, or diffusion (Yao et al. 1971). The attachment rate coefficient is related to the collision efficiency η and the sticking efficiency α as follows (Yao et al. 1971)

$$k_{att} = \frac{3}{2}\frac{(1-n)}{d_c}\alpha\eta\upsilon \qquad \text{[Eq. B2]}$$

Here, d_c is the average diameter of collision (grain size) in mm (Table B5). The fraction of particles that collide with the collector is given by η, the collision efficiency. Bacteria can be regarded as colloidal particles because of their size and surface charge. Bacteria are small in size and their transport in the immediate vicinity of the collector surface is dominated by Brownian diffusion, whereas effects of interception and gravitation are negligible. In this case the collision efficiency is given by the Smoluchowski-Levich approximation (Penrod et al. 1996):

$$\eta = 4A_S^{1/3}N_{Pe}^{-2/3} \qquad \text{[Eq. B3]}$$

Here, $N_{Pe} = d_c\eta\upsilon/D_{BM}$, a Péclet number, accounts for diffusion; $D_{BM} = K_B(T + 273)/(3\pi d_p\mu)$ is the diffusion coefficient, [$m^2\ s^{-1}$]; K_B = 1.38 × 10–23 is the Boltzmann constant [$J\ K^{-1}$]; T is temperature; d_p is the bacteria particle size; μ is the dynamic viscosity [$kg\ m^{-1}\ s^{-1}$]; $A_S = 2(1 - \gamma^5)/(2 - 3\gamma + 3\gamma^5 - 2\gamma^6)$ is Happel's porosity dependent parameter, with $\gamma = (1 - \eta)^{1/3}$. Colloid filtration theory predicts that virus removal, $\log(C/C_0)$ where C_0 is the influent concentration, is proportional to $v^{-2/3}$ (Yao et al. 1971).

The sticking efficiency, α, represents the fraction of the particles colliding with the solid grains that remain attached to the collector (Martin et al. 1992). The sticking efficiency reflects the net effect of repulsive and attractive forces between the surfaces of the particles and the collector and depends on the surface characteristics of the virus and soil particles. Therefore, it depends on pH, organic carbon content, and ionic strength. It is believed that α is independent of hydrodynamic effects (velocity and dispersion). Theoretical values of the sticking efficiency, and thus also of the attachment rate coefficient, considerably underestimate experimental values. For the purpose of modelling, we have chosen a typical value of $\alpha=10^{-4}$, following Schijven (1999).

APPENDIX C: MODEL OUTPUTS

SPASMO calculates a time series for plant growth and the uptake of nitrogen and phosphorus using a series of linked differential equations that are described in Appendix A. Growth and nutrient uptake will be species dependent, and so different input parameter values are used to distinguish pasture from short-rotation forest. In addition, growth and uptake are moderated by the soil's water and nutrient status. Any shortages at critical times over the growing season will have an impact on plant development.

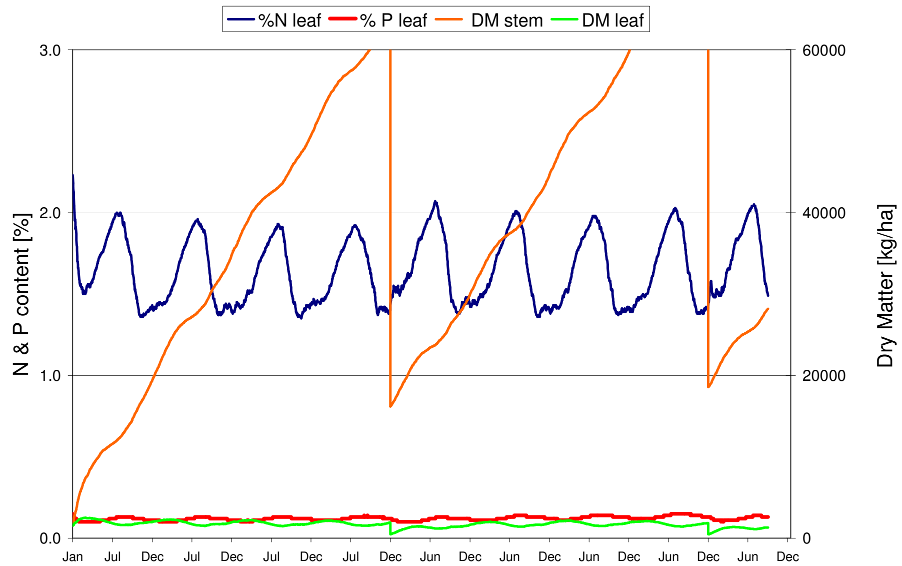

Figure C1. Time series of dry matter production and nutrient content for a short-rotation forest of *Eucalyptus ovata* trees managed under the preferred irrigation regime (Option 6) at the Homebush, Masterton site. Foliar N content varies between 1.5-2.1 mg kg^{-1}. N = nitrogen; P = phosphorus.

The time series of DM production and the N and P contents of leaf and stem material of *E. ovata* trees at the Homebush site are shown in Figures C1 and C2 for the case of irrigated and non-irrigated options, respectively. The trees were irrigated with 790 mm of effluent water that contained a total nutrient load of 91 kg-N ha^{-1} and 20 kg-P ha^{-1} each year. As a result they produced about 60 T ha^{-1} of stem DM every four years. Nutrient uptake and tree growth were approximately halved under the dry-land conditions (cf. Figures C1 and C2 and cf. Tables D1 and D7). Corresponding calculations for irrigated and dry-land pasture also showed a large drop in pasture production (cf. Figure C3 and C4) because of the low moisture and nutrient status of the soils under natural rain-fed conditions.

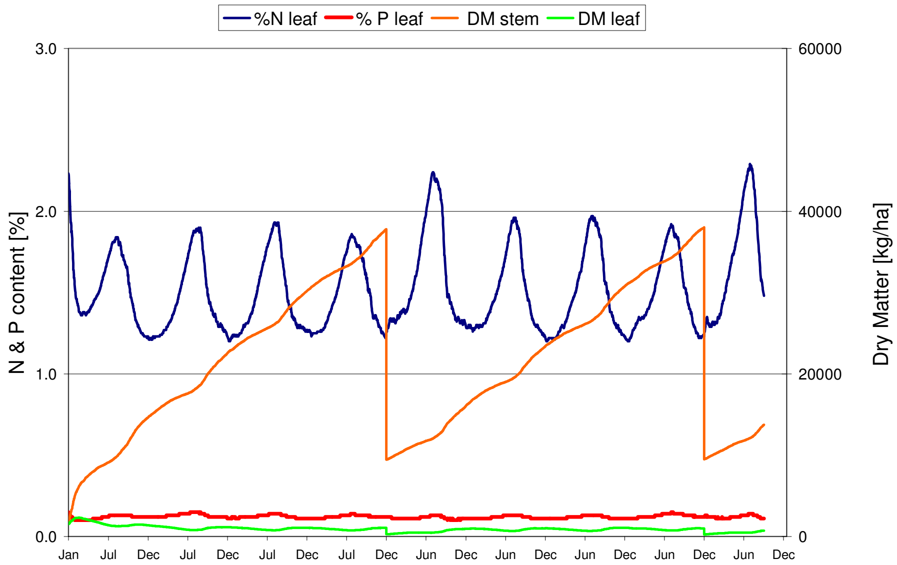

Figure C2. Time series of dry matter production and nutrient content for a short-rotation forest of *Eucalyptus ovata* trees managed under the preferred irrigation regime (Option 6) at the Homebush, Masterton site. Foliar N content varies between 1.5-2.1 mg kg^{-1}. N = nitrogen; P = phosphorus.

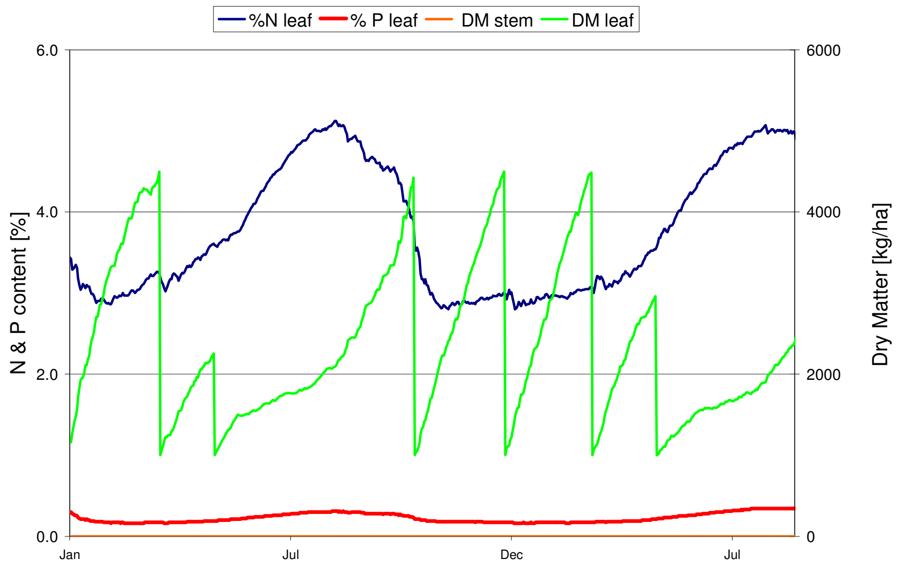

Figure C3. Time series of dry matter production and nutrient content for a cut-and-carry pasture (site 7) that is managed under the preferred irrigation regime (Option 6) at the Homebush, Masterton site. Foliar N content varies between 3.0 and 5.0%. The annual pasture production is about 12.2 T ha^{-1}. The cut-and-carry operation is predicted to remove some 340 kg-N ha^{-1} and some 29 kg-P ha^{-1} each year, on average. There would be 4-5 harvests each year. N = nitrogen; P = phosphorus.

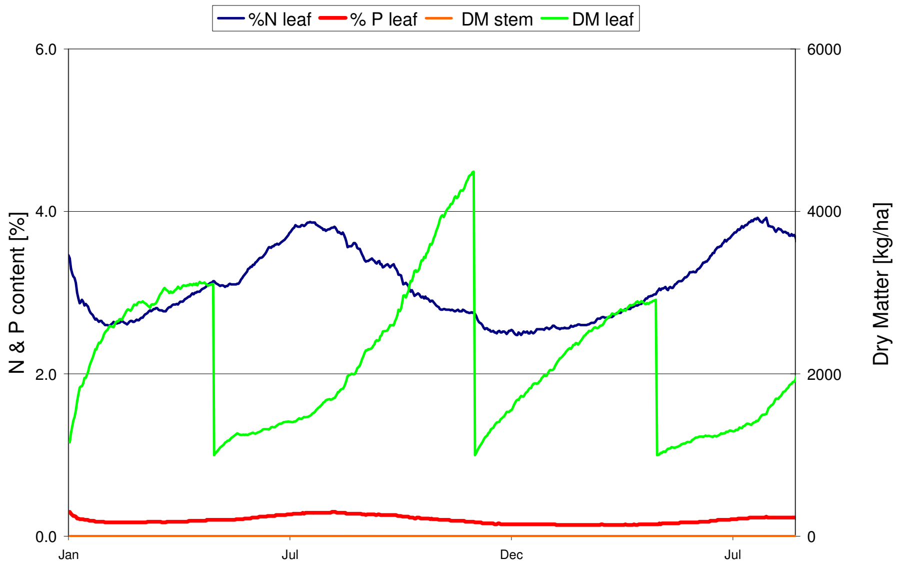

Figure C4. Time series of dry matter production and nutrient content for a cut-and-carry pasture at the Homebush, Masterton site (Zone 7) that is non-irrigated (the dry-land option). In this case the pasture is water and nitrogen limited. Foliar N content is very low and varies between 2.5 and 3.8%. The annual pasture production is only about 5.2 T ha^{-1}. In this case the cut-and-carry operation would remove just 91 kg-N ha^{-1} and just 5 kg-P ha^{-1} each year, on average. There would be 1-2 harvests each year. N = nitrogen; P = phosphorus.

SPASMO also calculates a time series for the depth-wise profile of water, nutrient concentrations (i.e. N and P) and contaminants (i.e. *E. coli* bacteria) in the soil under each site. Nutrients and contaminants either stay resident in the soil profile, or they degrade and dissipate, or they are carried along with drainage water that leaches through the soil profile. Figure C5 shows some examples of model predictions for the drainage water flux and the corresponding solution concentrations at 1.0 m that corresponds approximately to the depth of the ground water. We calculate a gradual rise over time for the dissolved reactive phosphorus (DRP) because progressively more P eventually gets leached through the profile. Nonetheless, the solution concentrations of P remain quite low (<20 µg L^{-1}) because of strong adsorption by the soil's clay particles. Nitrate-N concentrations show a strong seasonality, fluctuating between about 1.5 mg L^{-1} (winter) and 2.9 mg L^{-1} (summer). This is about 4 times lower than the NZDWS and therefore should not pose a significant risk to the quality of the local groundwater. The *E. coli* concentrations also show a strong seasonal pattern, with maximum values occurring in the late autumn (~ 60 cfu/100 ml) that are associated with increased effluent concentrations and the early onset of rains that exacerbate leaching losses. Additional dilution and die-off in the groundwater is expected to reduce the bacterial concentrations even further (Tate 1978).

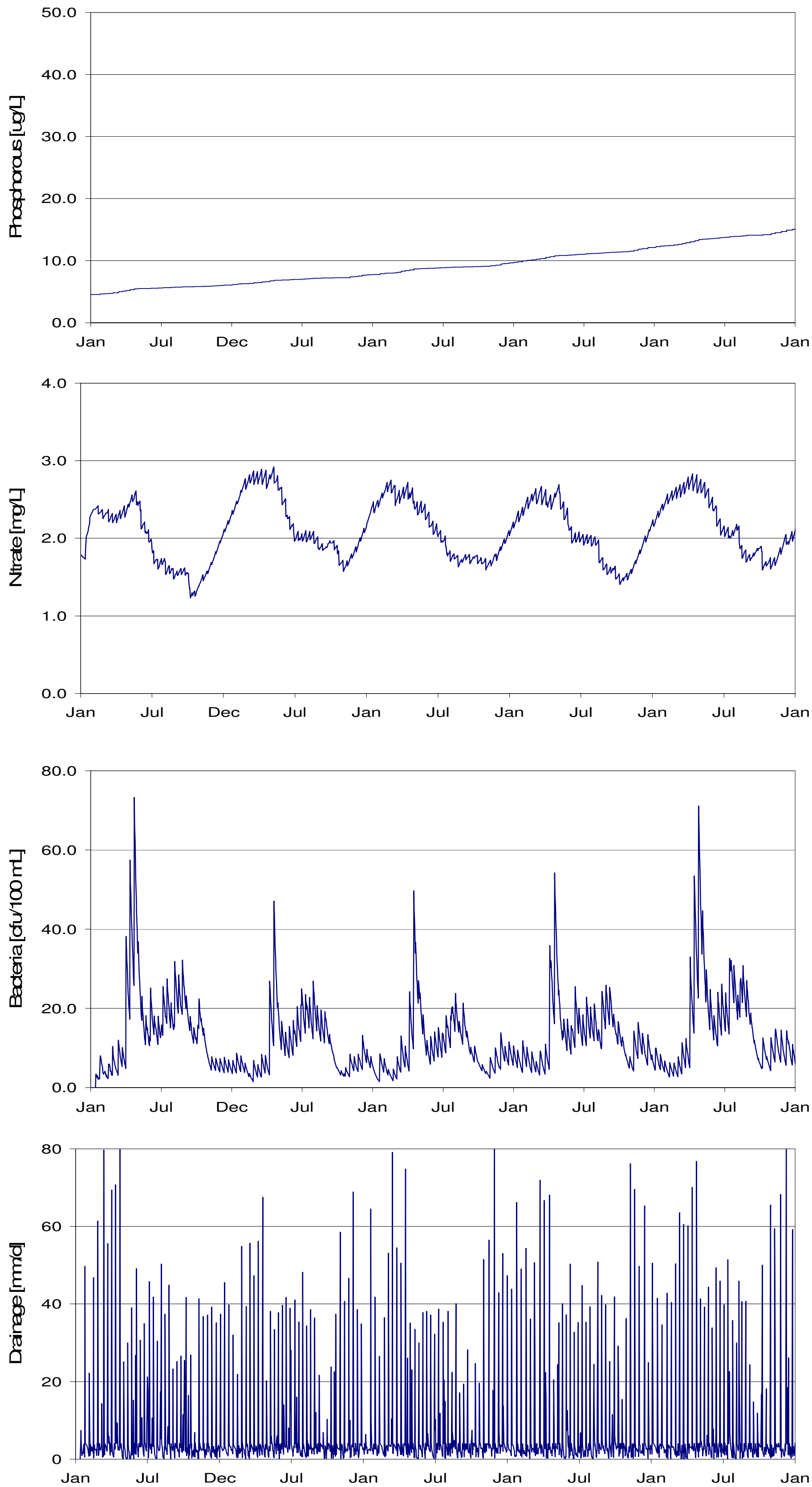

Figure C5. Time series showing predicted solution concentrations of phosphorus, nitrate-N and bacteria (i.e. E*scherichia coli*), and the corresponding drainage water fluxes under Zone 7 at the Homebush, Masterton site that operates under the preferred irrigation regime (Option 6).

APPENDIX D: STATISTICAL ANALYSIS OF MODEL PREDICTIONS

Table D1. Assessment of Environmental effects. Model output for the dry-land farm at the Homebush, Masterton site. The calculations are based on eight years of climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	623	613	609	621	609	632	632	613	612	612	743
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	0	0	0	0	0	0	0	0	0	0	0
Drainage	231	208	228	236	247	231	230	253	237	237	112
Runoff	12	24	28	2	5	2	0	0	0	0	26
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	0	0	0	0	0	0	0	0	0	0	0
Inactivity	0	0	0	0	0	0	0	0	0	0	0
Runoff	0	0	0	0	0	0	0	0	0	0	0
Resident	0	0	0	0	0	0	0	0	0	0	0
Leached	0	0	0	0	0	0	0	0	0	0	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ammonium [kg N/ha]											
Added	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Runoff	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Volatilized	5.6	6.9	6.0	5.6	6.9	5.6	5.6	6.9	6.9	6.9	5.2
Mineralized	78.5	80.6	81.8	78.0	80.7	77.4	78.0	82.9	82.7	82.7	71.9
Nitrified	-30.2	-30.9	-31.6	-30.0	-31.1	-29.9	-30.5	-32.1	-32.0	-32.0	-22.5
Resident	35.6	36.2	36.3	31.2	33.2	35.9	35.6	35.9	36.9	37.0	18.8
Leached	0.9	0.9	0.9	0.8	1.0	1.1	1.0	0.9	1.0	1.0	0.2
Conc. [mg/L]	0.2	0.2	0.2	0.2	0.2	0.3	0.2	0.2	0.2	0.2	0.1
Nitrate [kg N/ha]											
Added	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Runoff	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nitrified	30.2	30.9	31.6	30.0	31.1	29.9	30.5	32.1	32.0	32.0	22.5
Denitrified	2.0	2.9	3.2	2.6	3.5	0.7	1.6	4.1	2.1	2.1	0.2
Resident	3.2	3.5	3.3	3.4	3.1	2.6	2.8	3.0	3.3	3.2	1.0
Leached	15.4	14.5	14.8	14.2	14.7	17.0	16.5	15.0	16.8	16.8	4.1
Conc. [mg/L]	5.9	6.0	5.8	5.3	5.0	6.6	6.5	5.0	6.3	6.3	3.0
Phosphorus [kg N/ha]											
Added	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Runoff	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Resident	7.4	7.9	8.0	7.4	7.8	7.2	7.3	8.1	8.0	8.0	7.6
Mineralized	9.7	9.5	9.5	10.0	9.9	10.0	10.0	10.0	10.1	10.1	1.6
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Plant removal [kg/ha]											
DM removed	5297	5338	5408	5313	5316	5235	5235	5363	5335	5335	6813
N fixed	116.1	116.9	118.0	116.4	116.5	115.1	115.1	117.3	116.8	116.8	0.0
N removed	91.5	91.3	91.7	91.5	91.6	91.4	91.5	91.8	91.9	91.9	44.4
N topped	64.2	65.7	68.2	64.8	64.9	62.0	61.9	66.5	65.6	65.6	0.0
P removed	4.7	4.4	4.3	4.8	4.7	4.9	4.9	4.6	4.8	4.8	2.5
P topped	3.0	3.1	3.1	3.1	3.2	3.0	3.0	3.2	3.2	3.2	0.0

Table D2. Assessment of Environmental effects. Model output for Option 1 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1424	1528	1126	2231	1838	1975	2522	2524	2485	2485	789
Drainage	1358	1487	1041	2236	1819	1944	2537	2564	2500	2500	674
Runoff	108	57	126	26	53	50	6	2	8	9	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	71529	86098	56644	116992	92309	103380	132351	131118	129387	131096	26470
Inactivity	69861	84158	55025	115785	90508	101689	131319	129345	127970	129725	26468
Runoff	1285	1037	1264	297	988	950	188	140	358	297	0
Resident	3304	7807	3067	7579	6996	6361	6102	5664	5288	4937	0
Leached	0	12	0	44	16	12	179	802	379	420	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.1	0.0	0.0	0.3	1.8	0.9	1.0	0.0
Ammonium [kg N/ha]											
Added	81.9	87.8	64.7	128.3	105.7	113.5	145.0	145.1	142.9	142.9	45.4
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.3	0.1	0.5	0.5	0.0
Volatilized	18.1	15.1	17.1	19.6	15.3	19.3	20.0	16.9	15.6	15.6	20.0
Mineralized	80.5	82.9	83.9	80.0	82.6	79.4	80.7	85.1	84.8	84.8	71.5
Nitrified	-50.1	-56.2	-47.1	-71.5	-62.7	-61.0	-76.9	-83.9	-83.7	-83.6	-32.6
Resident	29.8	38.1	31.4	39.7	38.7	33.8	41.5	52.3	50.3	50.1	12.7
Leached	2.8	3.7	2.4	6.5	5.0	4.5	8.4	11.6	10.3	10.3	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.3	0.2	0.2	0.3	0.4	0.4	0.4	0.1
Nitrate [kg N/ha]											
Added	81.9	87.8	64.7	128.3	105.7	113.5	145.0	145.1	142.9	142.9	45.4
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.3	0.1	0.5	0.5	0.0
Nitrified	50.1	56.2	47.1	71.5	62.7	61.0	76.9	83.9	83.7	83.6	32.6
Denitrified	4.6	5.2	4.8	6.5	5.8	5.6	6.8	7.7	7.2	7.2	3.3
Resident	1.3	2.0	1.6	2.4	2.0	1.6	2.0	2.5	2.5	2.5	0.4
Leached	27.6	37.6	25.8	62.9	46.8	41.7	68.7	90.1	87.8	87.7	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.8	2.5	2.1	2.7	3.5	3.5	3.5	0.8
Phosphorus [kg N/ha]											
Added	35.6	38.2	28.1	55.8	46.0	49.4	63.0	63.1	62.1	62.1	19.7
Runoff	0.5	0.4	0.5	0.2	0.4	0.4	0.1	0.1	0.2	0.2	0.0
Resident	31.9	35.5	28.3	45.0	37.6	37.6	51.0	53.4	51.7	51.7	25.1
Mineralized	18.8	17.2	16.2	16.0	15.8	16.8	19.6	16.9	17.7	17.6	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.5	0.3	0.3	0.2
Plant removal [kg/ha]											
DM removed	10051	10074	9267	11545	10961	11409	12151	11743	11714	11691	12354
N fixed	165.5	165.7	156.8	174.3	170.9	173.2	180.1	176.1	175.8	175.6	0.0
N removed	246.8	249.3	202.2	335.1	319.8	334.7	340.7	335.1	336.1	336.2	90.5
N topped	83.9	85.4	95.6	51.0	44.7	46.2	74.9	58.9	57.8	57.4	0.0
P removed	18.5	17.1	13.6	25.9	24.2	27.7	28.6	25.8	27.2	27.1	4.5
P topped	6.7	5.9	6.0	3.8	3.3	3.6	6.0	4.4	4.5	4.5	0.0

Table D3. Assessment of Environmental effects. Model output for Option 2 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1424	1528	1130	2231	1838	1975	2980	3278	2926	2923	791
Drainage	1358	1487	1045	2236	1819	1944	2993	3309	2941	2937	676
Runoff	108	57	127	26	53	50	8	9	8	10	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	71529	86098	56729	116992	92309	103380	171842	176576	168598	169757	26504
Inactivity	69861	84158	55102	115785	90508	101689	169998	171870	165673	166760	26503
Runoff	1285	1037	1272	297	988	950	246	272	362	330	0
Resident	3304	7807	3066	7579	6996	6361	9152	11816	7900	7897	0
Leached	0	12	0	44	16	12	654	2515	1348	1427	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.1	0.0	0.0	1.1	5.0	2.8	2.9	0.0
Ammonium [kg N/ha]											
Added	81.9	87.8	65.0	128.3	105.7	113.5	171.3	188.5	168.2	168.1	45.5
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.4	0.5	0.5	0.6	0.0
Volatilized	18.1	15.1	17.1	19.6	15.3	19.3	20.5	22.8	15.8	15.8	20.1
Mineralized	80.5	82.9	83.9	80.0	82.6	79.4	81.3	85.9	85.2	85.2	71.5
Nitrified	-50.1	-56.2	-47.1	-71.5	-62.7	-61.0	-88.8	-99.8	-95.8	-95.7	-32.6
Resident	29.8	38.1	31.4	39.7	38.7	33.8	52.6	66.2	62.2	62.2	12.7
Leached	2.8	3.7	2.4	6.5	5.0	4.5	12.0	18.6	14.4	14.4	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.3	0.2	0.2	0.4	0.5	0.5	0.5	0.1
Nitrate [kg N/ha]											
Added	81.9	87.8	65.0	128.3	105.7	113.5	171.3	188.5	168.2	168.1	45.5
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.4	0.5	0.5	0.6	0.0
Nitrified	50.1	56.2	47.1	71.5	62.7	61.0	88.8	99.8	95.8	95.7	32.6
Denitrified	4.6	5.2	4.8	6.5	5.8	5.6	7.8	8.8	8.0	8.0	3.4
Resident	1.3	2.0	1.6	2.4	2.0	1.6	2.5	3.1	2.9	2.9	0.4
Leached	27.6	37.6	25.9	62.9	46.8	41.7	90.5	127.3	113.6	113.3	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.8	2.5	2.1	3.0	3.9	3.8	3.8	0.8
Phosphorus [kg N/ha]											
Added	35.6	38.2	28.2	55.8	46.0	49.4	74.5	81.9	73.2	73.1	19.8
Runoff	0.5	0.4	0.5	0.2	0.4	0.4	0.2	0.2	0.2	0.2	0.0
Resident	31.9	35.5	28.3	45.0	37.6	37.6	62.1	71.0	62.1	62.0	25.2
Mineralized	18.8	17.2	16.2	16.0	15.8	16.8	21.7	20.1	19.1	19.0	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.8	0.4	0.4	0.2
Plant removal [kg/ha]											
DM removed	10051	10074	9275	11545	10961	11409	12548	12488	12035	12009	12358
N fixed	165.5	165.7	156.9	174.3	170.9	173.2	184.8	184.1	179.1	178.8	0.0
N removed	246.8	249.3	202.3	335.1	319.8	334.7	351.2	344.5	343.8	344.2	90.5
N topped	83.9	85.4	95.8	51.0	44.7	46.2	89.6	86.6	68.8	68.0	0.0
P removed	18.5	17.1	13.6	25.9	24.2	27.7	29.5	27.3	28.0	28.0	4.5
P topped	6.7	5.9	6.0	3.8	3.3	3.6	7.5	6.7	5.5	5.5	0.0

Table D4. Assessment of Environmental effects. Model output for Option 3 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1990	2630	1716	3411	2846	2969	3822	3820	3793	3780	786
Drainage	1875	2574	1592	3407	2814	2921	3841	3861	3814	3802	671
Runoff	152	73	163	35	65	64	0	0	0	0	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	107774	149944	92054	178524	151893	163084	198318	199185	197289	194603	26413
Inactivity	104759	146854	88799	176683	149004	160428	196478	195728	194582	191879	26411
Runoff	2117	1830	2323	560	1731	1298	0	0	0	0	0
Resident	7761	10058	8078	9448	9334	11172	11385	8821	9838	9500	0
Leached	4	139	2	198	119	105	634	2178	1426	1481	0
Conc. [cfu/100 ml]	0.0	0.3	0.0	0.4	0.2	0.2	0.9	3.8	2.6	2.7	0.0
Ammonium [kg N/ha]											
Added	114.4	151.2	98.7	196.1	163.6	170.7	219.8	219.7	218.1	217.3	45.2
Runoff	1.8	1.6	2.0	0.8	1.5	1.3	0.0	0.0	0.0	0.0	0.0
Volatilized	25.1	21.3	21.9	28.6	21.5	27.8	29.1	22.2	22.1	22.1	19.9
Mineralized	80.6	83.6	83.9	82.3	84.3	81.4	81.7	86.7	86.3	86.3	71.5
Nitrified	-57.3	-78.2	-56.1	-95.9	-82.0	-79.2	-102.9	-111.8	-112.2	-111.9	-32.5
Resident	33.8	48.7	36.7	49.4	49.0	44.5	53.8	66.2	62.8	63.8	12.8
Leached	4.0	8.2	3.7	12.7	9.5	8.5	17.0	22.8	20.5	20.4	0.6
Conc. [mg/L]	0.2	0.3	0.2	0.4	0.3	0.3	0.4	0.5	0.5	0.5	0.1
Nitrate [kg N/ha]											
Added	114.4	151.2	98.7	196.1	163.6	170.7	219.8	219.7	218.1	217.3	45.2
Runoff	1.8	1.6	2.0	0.8	1.5	1.3	0.0	0.0	0.0	0.0	0.0
Nitrified	57.3	78.2	56.1	95.9	82.0	79.2	102.9	111.8	112.2	111.9	32.5
Denitrified	6.2	7.8	6.2	9.2	8.1	8.2	9.7	9.9	9.8	9.7	3.3
Resident	1.4	2.3	1.7	2.7	2.3	2.2	2.6	2.7	2.7	2.8	0.4
Leached	34.5	68.9	35.3	102.5	74.0	66.5	113.0	141.9	140.0	139.6	5.5
Conc. [mg/L]	1.8	2.6	2.2	3.0	2.6	2.3	3.0	3.7	3.7	3.7	0.8
Phosphorus [kg N/ha]											
Added	49.7	65.8	42.9	85.3	71.1	74.2	95.6	95.5	94.8	94.5	19.6
Runoff	0.8	0.7	0.9	0.3	0.6	0.6	0.0	0.0	0.0	0.0	0.0
Resident	38.3	55.4	37.3	73.4	59.6	61.2	75.6	82.7	80.0	79.7	25.1
Mineralized	16.0	18.0	15.3	23.1	21.8	24.0	20.3	22.3	22.0	21.8	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.6	0.4	0.4	0.2
Plant removal [kg/ha]											
DM removed	11508	12306	10625	13218	12770	13031	14143	13330	13461	13382	12351
N fixed	173.9	181.9	169.2	193.2	187.4	190.8	199.3	194.8	194.8	194.6	0.0
N removed	334.3	347.1	293.0	356.5	350.3	354.5	447.5	388.5	401.2	402.0	90.4
N topped	49.5	78.4	60.9	117.7	97.8	109.6	63.2	92.6	82.8	79.9	0.0
P removed	26.0	25.3	19.5	28.2	28.3	30.1	38.0	30.5	33.2	33.2	4.5
P topped	3.7	5.7	4.2	9.1	7.9	9.2	5.1	7.8	7.1	6.8	0.0

Table D5. Assessment of Environmental effects. Model output for Option 4 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1438	1538	1126	2430	1874	2045	2980	3278	2934	2923	791
Drainage	1371	1497	1042	2428	1854	2012	2993	3309	2949	2937	676
Runoff	109	58	126	33	54	52	8	9	7	10	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	71965	86801	56022	132618	93942	107234	171842	176576	170964	169757	26504
Inactivity	70284	84699	54585	130444	91983	105336	169998	171870	167784	166760	26503
Runoff	1286	1122	1083	803	1046	1033	246	272	285	330	0
Resident	3407	8477	3067	11116	7871	7423	9152	11816	10594	7897	0
Leached	0	13	0	96	18	16	654	2515	1367	1427	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.2	0.0	0.0	1.1	5.0	2.8	2.9	0.0
Ammonium [kg N/ha]											
Added	82.7	88.5	64.7	139.7	107.8	117.6	171.3	188.5	168.7	168.1	45.5
Runoff	1.1	1.0	0.9	0.7	0.9	1.0	0.4	0.5	0.4	0.6	0.0
Volatilized	18.2	15.1	17.1	20.0	15.4	19.5	20.5	22.8	15.8	15.8	20.1
Mineralized	80.6	82.9	83.8	80.3	82.4	79.5	81.3	85.9	85.2	85.2	71.5
Nitrified	-50.4	-56.4	-47.1	-76.3	-63.3	-62.3	-88.8	-99.8	-95.9	-95.7	-32.6
Resident	30.0	38.9	31.4	46.3	40.0	35.8	52.6	66.2	64.2	62.2	12.7
Leached	2.9	3.7	2.4	7.7	5.1	4.8	12.0	18.6	14.4	14.4	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.3	0.2	0.2	0.4	0.5	0.5	0.5	0.1
Nitrate [kg N/ha]											
Added	82.7	88.5	64.7	139.7	107.8	117.6	171.3	188.5	168.7	168.1	45.5
Runoff	1.1	1.0	0.9	0.7	0.9	1.0	0.4	0.5	0.4	0.6	0.0
Nitrified	50.4	56.4	47.1	76.3	63.3	62.3	88.8	99.8	95.9	95.7	32.6
Denitrified	4.7	5.3	4.8	6.9	5.8	5.8	7.8	8.8	8.1	8.0	3.4
Resident	1.3	2.0	1.6	2.6	1.9	1.6	2.5	3.1	3.0	2.9	0.4
Leached	28.0	37.9	25.9	71.7	47.9	43.7	90.5	127.3	113.7	113.3	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.9	2.6	2.1	3.0	3.9	3.8	3.8	0.8
Phosphorus [kg N/ha]											
Added	36.0	38.5	28.2	60.8	46.9	51.1	74.5	81.9	73.4	73.1	19.8
Runoff	0.5	0.4	0.4	0.3	0.4	0.4	0.2	0.2	0.2	0.2	0.0
Resident	32.3	35.8	28.4	49.8	37.3	39.1	62.1	71.0	62.3	62.0	25.2
Mineralized	19.0	17.3	16.3	16.9	15.0	17.4	21.7	20.1	19.1	19.0	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.8	0.4	0.4	0.2
Plant removal [kg/ha]											
DM removed	10103	10113	9248	11765	11112	11587	12548	12488	12025	12009	12358
N fixed	165.8	165.9	156.7	176.2	171.2	174.7	184.8	184.1	179.1	178.8	0.0
N removed	246.3	248.7	202.0	337.1	331.4	334.9	351.2	344.5	344.0	344.2	90.5
N topped	85.8	86.7	95.6	60.0	36.2	52.1	89.6	86.6	69.0	68.0	0.0
P removed	18.5	17.0	13.6	26.2	25.2	27.9	29.5	27.3	28.0	28.0	4.5
P topped	6.8	6.0	6.0	4.5	2.6	4.1	7.5	6.7	5.6	5.5	0.0

Table D6. Assessment of Environmental effects. Model output for Option 5 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1906	2293	1684	2510	2364	2414	3839	2513	2510	2536	788
Drainage	1798	2251	1565	2517	2350	2387	3847	2554	2532	2559	674
Runoff	146	58	158	23	47	44	13	0	0	0	83
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	101903	121026	90401	128363	120078	123484	177870	130402	130388	132120	26458
Inactivity	99553	119730	87661	127464	118864	122294	175557	129425	129506	131271	26457
Runoff	1607	501	2043	141	464	444	434	0	0	0	0
Resident	6410	6728	6024	6369	6384	6378	5956	6478	6491	5973	0
Leached	2	33	2	28	26	18	1348	230	141	166	0
Conc. [cfu/100 ml]	0.0	0.1	0.0	0.1	0.1	0.0	1.6	0.5	0.3	0.4	0.0
Ammonium [kg N/ha]											
Added	109.6	131.8	96.8	144.3	135.9	138.8	220.7	144.5	144.3	145.8	45.3
Runoff	1.5	0.8	1.8	0.1	0.5	0.5	0.7	0.0	0.0	0.0	0.0
Volatilized	24.9	20.8	21.8	26.7	20.9	26.5	27.6	21.1	21.0	21.1	20.0
Mineralized	80.4	83.2	83.9	80.9	83.6	80.4	81.3	85.8	85.6	85.6	71.5
Nitrified	-56.0	-70.3	-55.6	-72.0	-70.5	-66.6	-107.1	-76.9	-76.7	-77.3	-32.6
Resident	31.6	40.2	34.9	36.1	39.3	34.1	47.9	45.0	43.6	44.5	12.7
Leached	3.8	6.3	3.6	6.5	6.5	5.6	19.8	8.7	8.1	8.3	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.5	0.3	0.3	0.3	0.1
Nitrate [kg N/ha]											
Added	109.6	131.8	96.8	144.3	135.9	138.8	220.7	144.5	144.3	145.8	45.3
Runoff	1.5	0.8	1.8	0.1	0.5	0.5	0.7	0.0	0.0	0.0	0.0
Nitrified	56.0	70.3	55.6	72.0	70.5	66.6	107.1	76.9	76.7	77.3	32.6
Denitrified	6.0	7.0	6.1	7.3	7.1	6.9	9.1	7.7	7.6	7.7	3.3
Resident	1.4	2.0	1.6	2.1	1.9	1.7	2.5	2.0	2.0	2.0	0.4
Leached	33.0	55.9	34.6	58.9	54.4	46.7	127.5	68.1	67.1	68.1	5.6
Conc. [mg/L]	1.8	2.4	2.2	2.3	2.3	1.9	3.5	2.7	2.6	2.6	0.8
Phosphorus [kg N/ha]											
Added	47.7	57.3	42.1	62.7	59.1	60.3	96.0	62.8	62.8	63.4	19.7
Runoff	0.6	0.4	0.8	0.1	0.2	0.2	0.3	0.0	0.0	0.0	0.0
Resident	36.4	47.8	36.6	51.4	48.5	48.0	75.9	52.8	52.1	52.7	25.1
Mineralized	15.4	16.5	15.1	18.8	19.1	20.3	19.3	19.1	19.9	20.0	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.2	0.2	0.2	0.2
Plant removal [kg/ha]											
DM removed	11283	11907	10553	12280	12226	12343	13836	12252	12279	12299	12347
N fixed	172.2	177.7	168.5	181.4	180.8	182.2	197.1	181.1	181.4	181.7	0.0
N removed	334.2	337.4	291.9	341.4	339.1	341.9	443.8	340.0	341.1	342.1	90.4
N topped	42.3	66.5	59.3	79.8	77.6	82.2	52.5	79.6	80.1	80.2	0.0
P removed	25.9	24.5	19.4	27.1	27.0	28.9	37.8	26.8	28.0	28.1	4.5
P topped	3.1	4.6	4.1	5.9	6.0	6.6	4.1	6.0	6.3	6.3	0.0

Table D7. Assessment of Environmental effects. Model output for Option 6 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1424	1528	1126	2231	1838	1975	2522	2524	2485	2485	789
Drainage	1358	1487	1041	2236	1819	1944	2537	2564	2500	2500	674
Runoff	108	57	126	26	53	50	6	2	8	9	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	71529	86098	56644	116992	92309	103380	132351	131118	129387	131096	26470
Inactivity	69861	84158	55025	115785	90508	101689	131319	129345	127970	129725	26468
Runoff	1285	1037	1264	297	988	950	188	140	358	297	0
Resident	3304	7807	3067	7579	6996	6361	6102	5664	5288	4937	0
Leached	0	12	0	44	16	12	179	802	379	420	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.1	0.0	0.0	0.3	1.8	0.9	1.0	0.0
Ammonium [kg N/ha]											
Added	81.9	87.8	64.7	128.3	105.7	113.5	145.0	145.1	142.9	142.9	45.4
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.3	0.1	0.5	0.5	0.0
Volatilized	18.1	15.1	17.1	19.6	15.3	19.3	20.0	16.9	15.6	15.6	20.0
Mineralized	80.5	82.9	83.9	80.0	82.6	79.4	80.7	85.1	84.8	84.8	71.5
Nitrified	-50.1	-56.2	-47.1	-71.5	-62.7	-61.0	-76.9	-83.9	-83.7	-83.6	-32.6
Resident	29.8	38.1	31.4	39.7	38.7	33.8	41.5	52.3	50.3	50.1	12.7
Leached	2.8	3.7	2.4	6.5	5.0	4.5	8.4	11.6	10.3	10.3	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.3	0.2	0.2	0.3	0.4	0.4	0.4	0.1
Nitrate [kg N/ha]											
Added	81.9	87.8	64.7	128.3	105.7	113.5	145.0	145.1	142.9	142.9	45.4
Runoff	1.1	0.9	1.1	0.4	0.8	0.9	0.3	0.1	0.5	0.5	0.0
Nitrified	50.1	56.2	47.1	71.5	62.7	61.0	76.9	83.9	83.7	83.6	32.6
Denitrified	4.6	5.2	4.8	6.5	5.8	5.6	6.8	7.7	7.2	7.2	3.3
Resident	1.3	2.0	1.6	2.4	2.0	1.6	2.0	2.5	2.5	2.5	0.4
Leached	27.6	37.6	25.8	62.9	46.8	41.7	68.7	90.1	87.8	87.7	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.8	2.5	2.1	2.7	3.5	3.5	3.5	0.8
Phosphorus [kg N/ha]											
Added	35.6	38.2	28.1	55.8	46.0	49.4	63.0	63.1	62.1	62.1	19.7
Runoff	0.5	0.4	0.5	0.2	0.4	0.4	0.1	0.1	0.2	0.2	0.0
Resident	31.9	35.5	28.3	45.0	37.6	37.6	51.0	53.4	51.7	51.7	25.1
Mineralized	18.8	17.2	16.2	16.0	15.8	16.8	19.6	16.9	17.7	17.6	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.5	0.3	0.3	0.2
Plant removal [kg/ha]											
DM removed	10051	10074	9267	11545	10961	11409	12151	11743	11714	11691	12354
N fixed	165.5	165.7	156.8	174.3	170.9	173.2	180.1	176.1	175.8	175.6	0.0
N removed	246.8	249.3	202.2	335.1	319.8	334.7	340.7	335.1	336.1	336.2	90.5
N topped	83.9	85.4	95.6	51.0	44.7	46.2	74.9	58.9	57.8	57.4	0.0
P removed	18.5	17.1	13.6	25.9	24.2	27.7	28.6	25.8	27.2	27.1	4.5
P topped	6.7	5.9	6.0	3.8	3.3	3.6	6.0	4.4	4.5	4.5	0.0

Table D8. Assessment of Environmental effects. Model output for Option 7 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1433	1538	1130	2358	1859	2019	2716	2899	2662	2662	789
Drainage	1367	1497	1045	2359	1839	1986	2729	2936	2677	2676	674
Runoff	108	58	127	30	54	51	8	5	8	10	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	71856	86796	56736	127483	93416	106094	148468	151064	145088	146623	26481
Inactivity	70175	84699	55110	125734	91495	104263	147155	148426	143253	144816	26480
Runoff	1286	1117	1272	632	1040	1019	246	256	362	313	0
Resident	3407	8477	3066	9097	7586	6978	7326	7305	6343	5918	0
Leached	0	13	0	75	17	15	282	1285	624	678	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.2	0.0	0.0	0.5	2.8	1.4	1.5	0.0
Ammonium [kg N/ha]											
Added	82.4	88.5	65.0	135.6	106.9	116.1	156.2	166.7	153.1	153.0	45.4
Runoff	1.1	1.0	1.1	0.6	0.9	0.9	0.4	0.3	0.5	0.5	0.0
Volatilized	18.1	15.1	17.1	19.9	15.4	19.4	20.2	20.8	15.7	15.7	20.0
Mineralized	80.6	82.9	83.9	80.2	82.4	79.4	81.0	85.5	85.0	85.0	71.5
Nitrified	-50.3	-56.4	-47.1	-74.6	-63.0	-61.8	-81.6	-91.3	-88.4	-88.4	-32.6
Resident	30.0	38.9	31.4	43.1	39.6	35.0	45.8	57.5	54.9	54.7	12.7
Leached	2.8	3.7	2.4	7.2	5.1	4.7	9.7	14.6	11.8	11.8	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.3	0.2	0.2	0.3	0.5	0.4	0.4	0.1
Nitrate [kg N/ha]											
Added	82.4	88.5	65.0	135.6	106.9	116.1	156.2	166.7	153.1	153.0	45.4
Runoff	1.1	1.0	1.1	0.6	0.9	0.9	0.4	0.3	0.5	0.5	0.0
Nitrified	50.3	56.4	47.1	74.6	63.0	61.8	81.6	91.3	88.4	88.4	32.6
Denitrified	4.7	5.3	4.8	6.8	5.8	5.7	7.3	8.2	7.6	7.6	3.3
Resident	1.3	2.0	1.6	2.5	1.9	1.6	2.2	2.6	2.6	2.6	0.4
Leached	27.8	37.9	25.9	68.3	47.3	42.9	76.9	106.4	97.4	97.2	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.8	2.5	2.1	2.8	3.6	3.6	3.6	0.8
Phosphorus [kg N/ha]											
Added	35.8	38.5	28.3	59.0	46.5	50.5	67.9	72.5	66.6	66.5	19.7
Runoff	0.5	0.4	0.5	0.3	0.4	0.4	0.2	0.1	0.2	0.2	0.0
Resident	32.1	35.8	28.4	48.0	36.9	38.5	55.6	62.2	55.9	55.8	25.2
Mineralized	18.9	17.3	16.2	16.6	14.8	17.1	20.6	18.8	18.4	18.3	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.5	0.3	0.3	0.2
Plant removal [kg/ha]											
DM removed	10055	10113	9276	11674	11045	11531	12366	12191	11864	11827	12354
N fixed	165.9	165.9	156.9	175.5	171.1	174.1	182.6	180.3	177.3	177.0	0.0
N removed	248.3	248.7	202.3	337.3	332.9	334.7	345.8	339.5	338.4	339.5	90.5
N topped	83.3	86.7	95.9	56.0	33.4	50.1	82.2	75.6	63.9	62.4	0.0
P removed	18.6	17.0	13.6	26.2	25.3	27.8	29.1	26.7	27.5	27.5	4.5
P topped	6.6	6.0	6.0	4.2	2.4	3.9	6.7	5.8	5.1	5.0	0.0

Table D9. Assessment of Environmental effects. Model output for Option 8 at the Homebush, Masterton site. The calculations are based on complete climate and river flow records from 1997-2005. N = nitrogen; P = phosphorus; DM = dry matter.

Water balance [mm/year]											
Site number	1	2	3	4	5	6	7	8	9	10	11
Evaporation	814	814	814	814	814	814	814	814	814	814	895
Rainfall	892	892	892	892	892	892	892	892	892	892	892
Irrigation	1359	1429	1110	1729	1622	1665	1796	1782	1794	1789	791
Drainage	1296	1393	1026	1739	1611	1647	1818	1824	1817	1813	676
Runoff	104	52	126	21	44	37	0	0	0	0	84
Contaminants											
Bacteria [cfu/m^2 * 10^3]											
Added	67975	78309	55859	88774	83229	86338	93489	92598	92120	93448	26504
Inactivity	66460	76999	54326	88070	82070	85484	92984	91977	91630	92975	26503
Runoff	1181	677	1254	87	587	343	0	0	0	0	0
Resident	2879	5478	2406	5292	4926	4399	4270	3961	3695	3454	0
Leached	0	6	0	7	9	3	18	144	60	69	0
Conc. [cfu/100 ml]	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.2	0.2	0.0
Ammonium [kg N/ha]											
Added	78.1	82.2	63.8	99.4	93.2	95.7	103.3	102.5	103.1	102.9	45.5
Runoff	0.9	0.7	1.1	0.1	0.4	0.3	0.0	0.0	0.0	0.0	0.0
Volatilized	18.0	15.0	17.0	18.9	15.2	18.9	19.0	15.3	15.3	15.3	20.1
Mineralized	80.7	83.0	83.8	80.1	82.7	79.6	79.7	85.0	84.6	84.5	71.5
Nitrified	-49.1	-54.2	-46.8	-58.4	-57.5	-54.5	-58.2	-64.2	-64.1	-64.1	-32.6
Resident	28.7	34.7	30.5	31.7	34.2	29.5	32.0	40.3	38.6	38.3	12.7
Leached	2.7	3.3	2.3	3.9	3.9	3.4	4.1	5.6	5.3	5.3	0.6
Conc. [mg/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.3	0.1
Nitrate [kg N/ha]											
Added	78.1	82.2	63.8	99.4	93.2	95.7	103.3	102.5	103.1	102.9	45.5
Runoff	0.9	0.7	1.1	0.1	0.4	0.3	0.0	0.0	0.0	0.0	0.0
Nitrified	49.1	54.2	46.8	58.4	57.5	54.5	58.2	64.2	64.1	64.1	32.6
Denitrified	4.5	5.0	4.7	5.4	5.3	4.9	5.3	6.1	5.9	5.9	3.4
Resident	1.4	1.9	1.6	2.1	1.8	1.5	1.7	2.0	2.0	2.0	0.4
Leached	26.5	34.6	25.4	40.7	38.1	32.7	38.3	51.1	50.6	50.6	5.6
Conc. [mg/L]	2.0	2.4	2.4	2.3	2.3	1.9	2.1	2.8	2.7	2.7	0.8
Phosphorus [kg N/ha]											
Added	34.0	35.7	27.7	43.2	40.5	41.6	44.9	44.5	44.8	44.7	19.8
Runoff	0.4	0.3	0.5	0.0	0.2	0.1	0.0	0.0	0.0	0.0	0.0
Resident	31.1	34.1	27.9	36.4	34.8	34.1	33.9	39.2	37.8	37.7	25.2
Mineralized	19.1	17.5	16.1	17.2	16.8	18.3	15.4	17.2	16.9	16.7	3.0
Leached	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Conc. [ug/L]	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
Plant removal [kg/ha]											
DM removed	9822	9798	9230	10605	10535	10659	10975	10646	10733	10652	12358
N fixed	163.9	164.5	156.2	168.9	167.9	168.9	170.3	169.0	169.4	169.4	0.0
N removed	234.3	237.7	201.6	292.2	290.0	291.9	333.4	291.0	305.5	308.0	90.5
N topped	89.8	89.5	94.5	61.3	58.9	63.3	32.0	63.4	51.7	48.3	0.0
P removed	17.7	16.1	13.5	22.3	21.4	23.5	27.1	21.6	23.8	23.8	4.5
P topped	6.9	6.2	5.9	4.7	4.5	5.2	2.4	4.7	4.0	3.7	0.0