Predicting nitrogen inputs to Lake Rotorua using ROTAN-Annual

Prepared for Bay of Plenty Regional Council

October 2016

Prepared by:
Kit Rutherford

For any information regarding this report please contact:

Kit Rutherford
Principal Scientist
Catchment Processes
+64-7-856 1772
kit.rutherford@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
82 Ford Road
Napier 4110

Phone +64 6 834 1228

NIWA CLIENT REPORT No:	2016102HN
Report date:	October 2016
NIWA Project:	BOP16201

Quality Assurance Statement		
S. Yalden.	Reviewed by:	Sharlene Yalden
A. Bartley	Formatting checked by:	Alison Bartley
[illegible]	Approved for release by:	David Roper

Contents

Tables

Figures

Executive summary

This report describes application of the new ROTAN-Annual model that was developed to predict the effects of land use on nitrogen loads entering Lake Rotorua. Its annual time-step makes ROTAN-Annual consistent with OVERSEER and the target annual load for Lake Rotorua (435 t y^{-1}). This study complements previous predictions made using the weekly time-step model ROTAN-2011 and incorporates new groundwater boundaries, recent stream monitoring data and nitrogen losses calculated using an updated version of OVERSEER.

Predicted annual flows matched observations within 95% confidence limits. Groundwater from an 'extra' aquifer lying to the west and north of the surface catchment boundary is included in the water budget based on boundaries defined by GNS-Science.

In the model nitrogen travels to the lake by three pathways: 'quickflow' and 'streamflow' (that reach the lake within a year), and 'slowflow' (groundwater that may take decades to reach the lake). Nitrogen 'loss' along each pathway is quantified using separate attenuation coefficients. Catchment-scale attenuation (viz., the sum of all nitrogen losses from land and point sources minus the load entering lake) was estimated to lie in the range 32%-50%. This compares favourably with published estimates of catchment-scale attenuation in other catchments both in New Zealand and overseas.

Several different combinations of quickflow, slowflow and streamflow attenuation coefficients gave similar fits between observed and predicted stream nitrogen concentrations. This occurred because ROTAN-Annual is 'over parameterised' in that the three attenuation coefficients were calibrated using only stream nitrogen concentrations, albeit measured at ten different monitoring sites. There is no way of deciding solely by examining calibration results which combinations of attenuation coefficients are truly optimal. Were one or more of the attenuation coefficients to be quantified independently then the other attenuation coefficient(s), and the steady-state lake load, could be determined more precisely. This was attempted in the study using a small number of groundwater measurements to estimate the slowflow attenuation coefficient.

Spatially uniform attenuation coefficients did not produce a satisfactory match between observed and predicted TN concentrations at all monitoring sites, while spatially variable coefficients differed between catchments without any obvious reasons. Either there are errors in the model inputs (nitrogen losses, groundwater flow pathways and travel times), and/or there are errors in the stream monitoring data used for calibration, and/or there are spatial differences in attenuation.

Two published estimates of groundwater residence time gave similar (within 4%) predictions of steady-state lake load. However, were groundwater flow pathways and/or travel times to differ significantly from those modelled, then calibrated attenuation coefficients and steady-state lake loads could change.

Predicted nitrogen concentrations and the steady-state lake load was insensitive to quickflow attenuation because high drainage in the catchment means the little nitrogen travels via the quickflow pathway. The calibrated slowflow and streamflow attenuation coefficients were inversely correlated – as expected. Nevertheless, the steady-state lake load was negatively correlated with both the slowflow and streamflow attenuation coefficients.

The model was used to predict the steady-state lake load assuming that nitrogen losses remain at current levels. It was also used to predict the steady-state lake load assuming nitrogen losses are

reduced as specified by Bay of Plenty Regional Council (BoPRC), together with the rate at which lake loads are expected to decrease.

Frequency distributions of steady-state lake load were determined: after calibrating the model to observed stream nitrogen concentrations, and then using the results of synoptic groundwater sampling to constrain the slowflow attenuation coefficient. These frequency distributions are only approximate because the 'true' distributions of uncertainty in nitrogen losses, groundwater travel times and groundwater pathways are unknown.

The 'most likely' steady-state lake load assuming current losses and excluding rainfall on the lake was estimated to be 750 t y^{-1}, with 95% confidence limits of 670-840 t y^{-1}. This range encompasses the value of 725 t y^{-1} estimated using ROTAN-2011.

The 'most likely' steady-state lake load assuming nitrogen reductions and again excluding rainfall on the lake was estimated to be 420 t y^{-1}, with 95% confidence limits of 390-460 t y^{-1}. The target load of 405 t y^{-1} falls within this range. While the 'most likely' lake load (420 t y^{-1}) exceeds the target load (405 t y^{-1}), the difference may not be statistically significant. Nevertheless, while the nitrogen loss reductions specified by BoPRC are predicted to significantly reduce lake loads, they may not quite reach the target load. One reason is that the target reductions for engineering and gorse made up 25% of total reductions when set in 2011 but, because of the changes to OVERSEER, make up only 15% in this study.

ROTAN-Annual predicts lake loads to drop at a slightly slower rate than the 2011 study. Nevertheless, the nitrogen reductions specified by BoPRC are expected to reduce lake loads to within 25% of the target after 25 years although steady-state may not be reached until after 2100.

1 Background

Excess nitrogen inputs to Lake Rotorua were identified as a cause of algal growth and impaired lake water quality (Environment Bay of Plenty 2007). As a part of a lake improvement strategy, Bay of Plenty Regional Council (BoPRC) have investigated several mitigation strategies focused on reducing the nitrogen and phosphorus mass load entering the lake. Reducing the input of nutrients will gradually lead to a reduction in the growth rate of phytoplankton, which will in turn reduce the average phytoplankton biomass and the frequency of 'blooms', increase water clarity and reduce the frequency and severity of oxygen depletion in the bottom waters.

The ROTAN-2011 model (Rutherford et al. 2008, 2009, and 2011) predicts the effects of land use changes on nitrogen loads delivered to Lake Rotorua in surface and groundwater at daily or weekly intervals. It was used to help set sector loads for nitrogen required to not exceed the target input for Lake Rotorua of 435 t y^{-1}. The original ROTAN-2011 model relied on OVERSEER v5 to estimate nitrogen losses from farmland. That version of OVERSEER is no longer supported and has been replaced by OVERSEER v6. For the same inputs, OVERSEER v6 predicts significantly higher nitrogen losses from farms in the Rotorua catchment than v5 (Alastair MacCormick, pers. comm.).

BoPRC intended using OVERSEER v6 and ROTAN-2011 to help manage nitrogen within the Lake Rotorua watershed. However, the original ROTAN-2011 needed to be re-calibrated before it could be used with OVERSEER v6. Recalibration of ROTAN-2011 proved impossible within the time available because of a requirement for extensive reprogramming. This was necessitated following upgrades made by ESRI to the ArcGIS software, and by Microsoft to Visual Basic and Microsoft Access components. Reprogramming of ROTAN-V2 is currently being undertaken by the University of Waikato to provide daily-weekly nitrogen loads for their lake water quality models, but that work is incomplete to-date.

As a consequence of ongoing delays in the redevelopment of ROTAN-2011 (version hereafter referred to as ROTAN-V2), NIWA was commissioned by BoPRC in March 2016 to develop a simplified version (hereafter referred to as ROTAN-Annual) and calibrate it using:

- OVERSEER v6 nitrogen losses
- recent stream monitoring data
- revised groundwater boundaries, and
- updated land use information.

It was envisaged that ROTAN-Annual would provide technical support to BoPRC and stakeholders during the plan change process.

2 Methods

2.1 Model structure

ROTAN-Annual builds on modelling work undertaken in Hawke's Bay during 2010-2011 (Rutherford 2013). For that study, an annual time-step nitrogen and phosphorus sub-model (TRIM-Catchment) was developed as a component of the TRIM (Tukituki River Model) nutrient-periphyton model. The concepts of TRIM-Catchment have been further developed and the model re-coded during this study, to produce ROTAN-Annual. Major modifications include new calibration and sensitivity assessment routines. ROTAN-Annual operates with a time-step of one year over the period 1900-2050 and has been calibrated over the period 1975-2015.

2.2 Watershed boundaries

White et al. (2014) recently re-defined the surface catchment boundary of Lake Rotorua[1]. Water balance studies had indicated that the groundwater catchment extends beyond the surface catchment (White et al. 2014, Rutherford et al. 2008), notably north-west of the lake on the Mamaku Plateau. White et al. (2014) also re-defined the outer boundary of the groundwater catchment that drains to the lake[2].

In ROTAN-Annual, sub-catchment surface boundaries and the streamflow network were defined using the River Environment Classification (REC)[3] 'clipped' by the GNS outer groundwater boundary (Figure 2-1). The REC sub-divides the Lake Rotorua watershed into c. 240 sub-catchments. To better model groundwater, some of the larger sub-catchments (notably elongated sub-catchments near Awahou, Hamurana and Ngongotaha) were split into smaller sub-catchments resulting in 280 sub-catchments in total. In the 'extra' sub-catchments (i.e., those which lie inside the outer groundwater boundary but outside the surface catchment of the lake), the REC streams flow away from the lake – streamflows (and associated nitrogen loads) from these sub-catchments were not included in the water or nitrogen budgets for the lake, but groundwater was assumed to reach the lake.

[1] GNS file: Lake_Rotorua_SC_1_2000_12_May_2014
[2] GNS file: Lake_Rotorua_GW_1_2000_ 9_June_2014
[3] https://data.mfe.govt.nz/ data/category/fresh-water

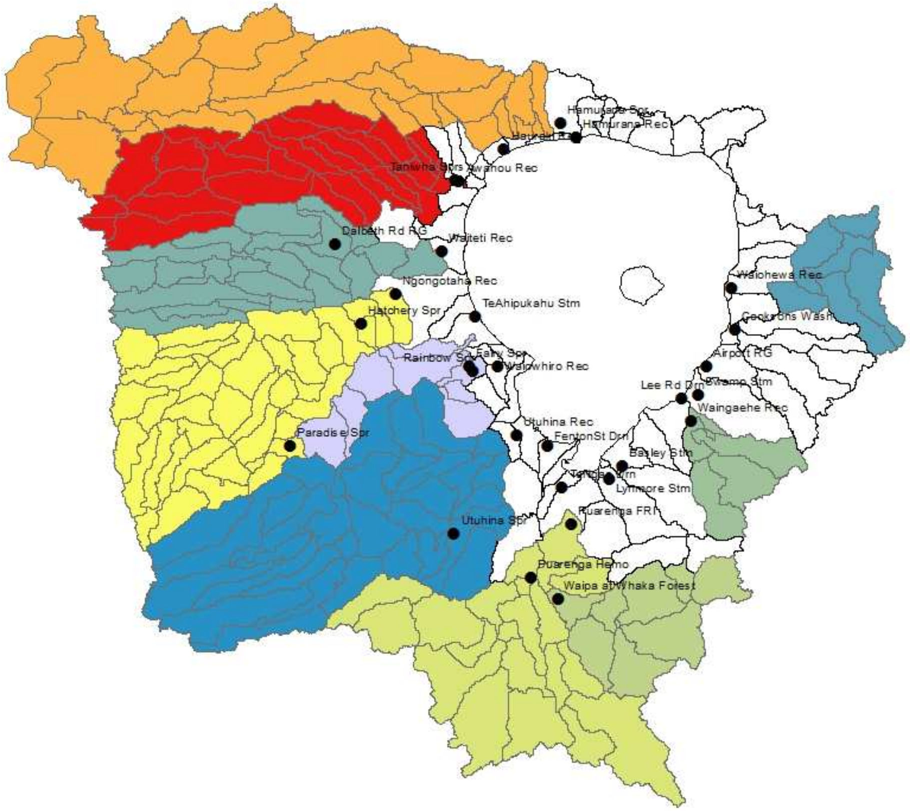


Figure 2-1: The outer boundary of the Rotorua watershed as determined by GNS. Also shown are the 280 sub-catchments used for modelling which are based on the REC. The colouring denotes aquifers that drain to a streamflow recorder. Key monitoring sites for flow and nitrogen are indicated by black dots.

2.3 Rainfall and total runoff

The spatial distribution of annual rainfall across the watershed was quantified using the 30-year (1981-2010) average rainfall isopleths supplied as an ArcGIS layer (Alastair MacCormick, pers. comm.) (Figure 2-2). Annual rainfall time-series at two reference gauges (Rotorua Airport and Ngongotaha at Dalbeth Road or Okere Falls) were extracted from the NIWA climate database (CliDB) for the period 1920-2015. In each REC sub-catchment a rainfall scaling factor was calculated as the average in the sub-catchment divided by the average at the two reference gauges. Rainfall scaling factors were assumed constant over time. In each year of the simulations, annual rainfall in each sub-catchment was calculated by scaling the average rainfall at the two reference gauges. Prior to 1920, synthetic annual rainfall time-series were generated by repeating the 1920-1939 time series.

Total annual runoff from each sub-catchment is the difference between annual rainfall (RAIN) and annual actual evapotranspiration (AET). Daily potential evapotranspiration (PET) was calculated from meteorological data measured at Rotorua Airport and summed to provide annual totals. Three different formulae were used (Penmann-Monteith for pasture, Penmann-Monteith (modified) for forest, and Jobson (1975) for open water) as detailed in Rutherford et al. (2008). Annual AET was estimated from rainfall (RAIN) and PET using the empirical model of Zhang et al. (2001).

$$\frac{AET}{RAIN} = (1 + w\frac{PET}{RAIN}) / (1 + w\frac{PET}{RAIN} + \frac{RAIN}{PET}) \qquad 1$$

where *w* = the plant available water coefficient (dimensionless). Note that *w* is smaller for pasture than forest (i.e., forest has higher AET than pasture for the same rainfall (e.g., Rutherford et al. 2008, Figure 3)). The values of *w* for forest and pasture were estimated by matching observed and predicted average flow in the Ohau Channel while keeping the difference in *w* between pasture and forest at 0.1 based on values in Zhang et al. (2001).

2.4 Quickflow and slowflow separation

In ROTAN-Annual runoff and associated nitrogen loads are routed through the catchment as either quickflow or slowflow, and streamflow.

It is assumed that a percentage of runoff reaches the stream in the sub-catchment where it is generated in the year it is generated (hereafter termed 'quickflow'). Quickflow includes overland flow (which occurs infrequently in the catchment) and shallow sub-surface flow. In ROTAN-Annual quickflow carries the same proportions of water and nitrogen.

The complement of quickflow is 'slowflow' (i.e., water routed to streams via groundwater).

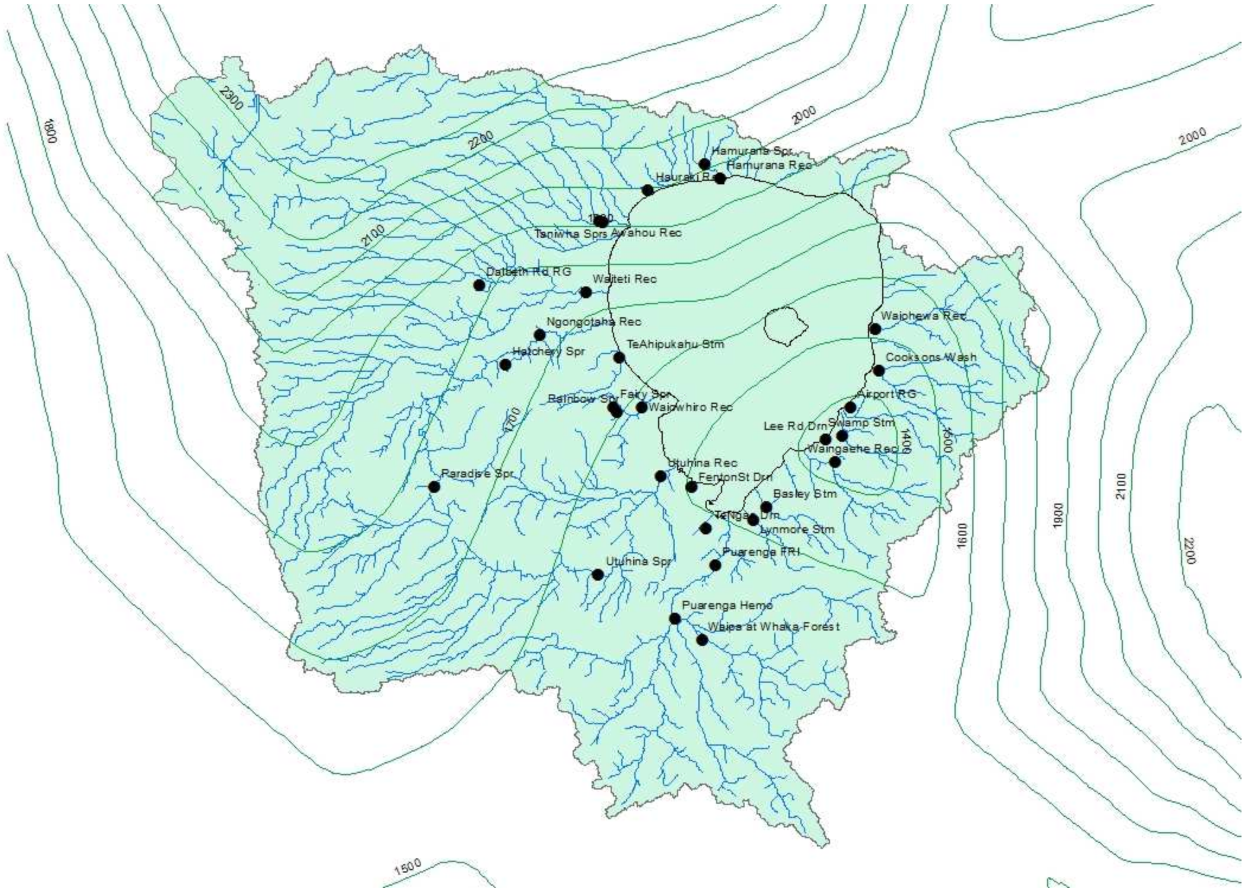


Figure 2-2: Rainfall contours (30 year average 1981-2010). Also shown are the REC streams, monitoring sites (black dots) and the outer boundary of the watershed. Note that in the northwest some streams flow away from the lake, but drainage from these sub-catchments was assumed to reach the lake.

2.5 Groundwater

Slowflow drains to groundwater and may re-emerge as springflow in a sub-catchment distant from where it drained. Nitrogen in slowflow may take years-decades to reach the lake (Morgenstern & Gordon 2006).

The ratio of slowflow/runoff was determined through baseflow separation of daily average flows measured in the major streams. Standard baseflow separation methods furnish estimates of slowflow that vary over days-weeks – ROTAN-Annual requires annual estimates. Annual slowflow was calculated as the average of the minimum stream flows in each four-month period. Hereafter the ratio of slowflow/runoff is called the 'drainage fraction' (viz., the proportion of total runoff that drains to an aquifer and becomes slowflow, expressed as a percentage). The drainage fraction (symbol *D*) was assumed to be spatially homogeneous across all land draining to a flow recorder (viz., spatially uniform above each aquifer). However, the drainage fraction varied between aquifers (viz., between flow recorders).

ROTAN-Annual assumes that drainage from each REC sub-catchment flows through a streamtube to either a stream segment or the lake. Internal aquifer boundaries define the REC sub-catchments that drain to a particular location. Preliminary boundaries for each aquifer were defined using previous estimates (White & Rutherford 2009) and the drainage fraction specified. ROTAN-Annual was then run and REC sub-catchments along the boundary of the aquifer were either included in, or excluded from, the aquifer (by trial and error) until observed and predicted flows matched at the flow recorder site. Ungauged catchments were assigned a default drainage fraction of 70%.

Slowflow along each streamtube is modelled in ROTAN-Annual using an exponential piston flow model (EPM) similar to that used by Morgenstern et al. (2004, 2015). Drainage enters a well-mixed linear reservoir whose outflow is proportional to the volume of water above a specified outlet height (hereafter termed the exponential (E) component). The volume of water above the outlet height is

$$V_1^{n+1} = V_1^n \exp(-\rho\Delta t) + \frac{q_{in}^n}{\rho}(1 - \exp(-\rho\Delta t)) \qquad 2$$

$$q_{out}^n = \rho V_1^n \qquad 3$$

where *n* = year number, V_1^n = annual average volume in year *n* (m^3), q_{in}^n = annual inflow in year *n* (viz., sub-catchment drainage) ($m^3\ y^{-1}$), q_{out}^n = annual outflow from the reservoir ($m^3\ y^{-1}$), Δt = time-step (1 year) and ρ = reservoir outflow coefficient (y^{-1}). Decreasing ρ reduces the year-to-year variations in spring flow arising from variations in rainfall. The value of ρ was assumed uniform in each internal aquifer but varied between aquifers. Its values were calibrated to match observed and predicted year-to-year variations in annual flow at the ten recorder sites.

Reservoir outflow travels as piston flow to an upwelling location (viz., a spring or the lake) (hereafter termed the piston flow (P) component). The proportion of mean residence time associated with the E and P components was specified based on the fractions determined by Morgenstern et al. (2004, 2015) (see Table 2-1). For each year inflow equals outflow for the P pathway (viz., there is no flow 'lag' associated with the P component, although the E component 'smooths' variations in annual flow). Note that the EPM model is a conceptual model and the variables *V* and q_{out}, together with the piston flow travel time, have no physical basis: although the resulting mean residence time (MRT) can be measured using tracers (e.g., Morgenstern et al. 2005).

2.6 Flow calibration

Flow calibration involved estimating three coefficients

- the plant available water coefficient *w* (Eq. 1)
- the drainage fraction *D*, and
- the reservoir outflow coefficient *ρ* (Eq. 2).

In summary:

- The plant available water coefficient was calibrated so that observed and predicted flows in the Ohau Channel matched, while keeping the difference in *w* between pasture and forest at 0.1.
- Drainage fractions in each aquifer were determined through baseflow separation in the major streams.
- The reservoir outflow coefficient in each aquifer was calibrated so that observed and predicted year-to-year variations in annual flow matched (assessed visually).
- Internal aquifer boundaries were adjusted so that the average observed and predicted flows in the major streams matched within the 95% confidence limits of the observed flows.

2.7 Forms of nitrogen

Nitrogen concentrations in streams flowing into Lake Rotorua have been monitored intermittently since the late 1960s. The early studies (Fish 1975, Hoare 1980b, Williamson et al. 1996) report nitrate, ammonium and nitrite concentrations – which sum to dissolved inorganic nitrogen (DIN) – but only a few particulate (PN), dissolved organic (DON) or total nitrogen (TN) concentrations. Although BoPRC included TN in their stream monitoring commencing in the 1990s, TN data are sparse prior to 2000.

In some streams the time trends of DIN and TN are similar. In other streams, however, the time trends differ – notably during the 1990s. This may reflect problems with accurately measuring TN in the laboratory during the 1990s, or they may reflect land use changes (e.g., the Kaituna Catchment Control Scheme).

It is not clear from the OVERSEER documentation what forms of nitrogen it includes and what forms it omits. It is acknowledged that dissolved organic nitrogen (DON) is not included, even though soils leach DON. Flood losses are not included and so particulate nitrogen (PN) losses are likely to be underestimated. Nitrogen losses from urine patches are a major contributor to losses from dairy farms and these are predominantly in DIN form.

The question arises whether ROTAN-Annual should be calibrated to observed DIN or TN concentrations. DIN data extend back to the 1970s and so the model's ability to predict time trends can be assessed using DIN. However, the model will be used to examine management strategies to achieve the target lake inflow of 435 t y^{-1} – this target relates to TN not DIN (Environment Bay of Plenty 2007, 2009). Consequently, in this study ROTAN-Annual was calibrated to annual flow-weighted average TN concentrations measured in the major streams.

2.8 Nitrogen losses

The ROTAN-2011 study estimated nitrogen losses using OVERSEER v5 assuming 'generic' or 'typical' dairy and drystock farms (Rutherford et al. 2010, 2011). For this study BoPRC re-examined historic land use maps and used OVERSEER v6.2.0 to re-estimate nitrogen losses (Alastair MacCormick, BoPRC, pers. comm.). Methods are detailed elsewhere (MacCormick 2016) and a brief outline is given here.

Soil types (defined in S-map) and rainfall band (defined by the 1981-2010 average 200 mm rainfall contours) were intersected giving 82 polygons which were used to help define blocks in OVERSEER. There is insufficient information to model farm management in detail prior to 2001 (e.g., seasonal variations in stocking rate, differences in stocking rates and differences in fertiliser use are unknown). However, for the years 1940, 1958, 1974, 1986 and 1996, information was gathered about 'typical' (generic) dairy and drystock farms from agricultural statistics and discussions with an 'expert panel', as detailed in Rutherford et al. (2010). This information was reviewed as part of the current study through further discussions involving farmers with knowledge of the history of land use and farm management in the catchment. One difference from the ROTAN-2011 study was to model all farms from 1940-1958 as 'mixed' with a mixture of dairy, drystock and cropping. BoPRC then re-modelled 'generic' mixed, dairy and drystock farms from 1940-2003 using OVERSEER v6.

In 2001-2004 (assigned to 2003) BoPRC collected 'benchmarking' data which enabled each farm, and blocks within farms, to be modelled in detail. This data was updated in 2015 where changes in land use occurred. No consistent patterns could be found between relative productivity and soils, rainfall, elevation or slope and, consequently, variability between farms was assumed to be random across the watershed.

For 2003 BoPRC re-modelled farms using both 'generic' data (from the earlier study) and 'benchmarking' data (more detailed). OVERSEER v6 was used in both cases and results for each block were compared (viz., a pairwise comparison was made between losses from farms with similar rainfall, soils and enterprise). On average 'benchmarking' losses were 28% and 39% higher for dairy and drystock farming respectively (dairy 128 ± 11%, drystock 139 ± 9%, mean ± 95% confidence interval). There are several possible reasons for these differences:

1. The 'generic' modelling assumed 50:50 sheep:beef on drystock farms whereas the benchmarking data indicates a higher proportion of cattle (e.g., dairy replacements).
2. The 'generic' modelling did not include cropping but the benchmarking data indicate 3-4% of farmland is cropped.
3. The 'generic' modelling did not model seasonal variations in stocking whereas the benchmarking data indicate significant over-wintering of dairy cattle on drystock farms in the catchment.
4. The 'generic' modelling assumed 'standard practice' on all farms whereas 'benchmarking' captures differences in farming practice between farms.

Although losses from farms from 2001-2015 can be modelled using the detailed benchmarking data, equivalent information is not available prior to 2003. For ROTAN-Annual, losses were re-calculated in OVERSEER v6 using 'generic' farm systems data for 1940, 1958, 1974, 1986 and 1996 and then scaled by 128% (dairy), 139% (drystock) and 133% (mixed) – the scaling factors S_i in Equations 7 and 8. Note that changes over time in stocking rates, animal types, fertiliser use etc., were captured in the input data used to model 'generic' farms.

2.9 Uncertainty in nitrogen loss rates

Three estimates of nitrogen loss were used in the modelling (low, mid and high) to account for uncertainty in the OVERSEER estimates. Variability and uncertainty arise from four sources.

First, it is reported that OVERSEER estimates of nitrogen loss fall within ±30% of measurements at test sites around New Zealand (Selbie et al. 2003). For 2003 and 2015, lower and upper bound losses from each land parcel (OVERSEER block) were estimated to be

$$L_i = O_i\,(1 + V_o RAND(-1{,}0)) \qquad 4$$

$$U_i = O_i\,(1 + V_o RAND(0{,}1)) \qquad 5$$

where L_i and U_i = lower and upper bound losses from land parcel *i* (kg ha^{-1} y^{-1}), O_i = predicted OVERSEER loss for the land parcel (given detailed 'benchmarking' data), V_o = uncertainty in OVERSEER estimates and *RAND* denotes a random number within the specified range.

Second, rainfall and soils affect nitrogen losses where farming enterprise and practice are identical. OVERSEER explicitly accounted for differences in soil and rainfall and no additional uncertainty was included.

Third, nitrogen losses vary with farming practice (notably stocking rate and drystock/dairy ratio). OVERSEER explicitly accounted for such differences for 2003 and 2015 because 'benchmarking' data gave an accurate picture of this source of variation. Prior to 2003 'typical' dairy, drystock or mixed farms were modelled using 'most likely', 'upper' and 'lower' bound stocking rates and drystock/dairy ratios estimated from farming statistics and expert opinion.

Fourth, BoPRC analysis of the 2003 benchmarking data showed significant variations between farms of the same type, in the same rainfall band and on similar soils. This variation is not captured in the modelling of 'generic' farms. However, benchmarking losses for 2003 were grouped by rainfall and soil type and an approximate analysis was made of the variations between farms

$$V_f = \frac{1}{2n}\sum \frac{max - min}{average} \qquad 6$$

where V_f = variation between farms of the same type, *max*, *min* and *average* = maximum, minimum and average losses in each rainfall/soil type group (kg ha^{-1} y^{-1}) and *n* = number of groups. For dairy and drystock farming the variations were ±25% and ±30% respectively. These figures imply that for the same type of farm with the same soils, rainfall and stocking rate, the 'best' farms lose 25-30% less, while the 'worst' farms lose 25-30% more, than the average.

For 1940, 1958, 1974, 1986 and 1996 lower and upper bound losses were estimated to be:

$$L_i = O_i\, S_i(1 + V_t\, RAND(-1{,}0)) \qquad 7$$

$$U_i = O_i\, S_i(1 + V_t\, RAND(0{,}1)) \qquad 8$$

$$V_t = \sqrt{{V_o}^2 + {V_f}^2} \qquad 9$$

where L_i and U_i = lower and upper bound losses from land parcel *i* (kg ha^{-1} y^{-1}), O_i = predicted OVERSEER loss for the land parcel (given 'generic' data), S_i = scaling factor (based on the comparison of 2003 benchmarking with 2003 generic losses), V_t = combined uncertainty for farms and OVERSEER, V_o = uncertainty in OVERSEER estimates, V_f = variation between farms, and *RAND* denotes a random number within the specified range.

2.10 Nitrogen in groundwater

The total nitrogen loss was sub-divided between quickflow and slowflow using the drainage factors described in Section 0. As described above, drainage was assumed to flow via a well-mixed reservoir (E) and a piston flow (P) pathway (the EPM model) before re-emerging in a stream or the lake. Slowflow nitrogen was routed through the reservoir (E component) using

$$M^{n+1} = M^n e^{-(q/V+\sigma)\Delta t} + m_{in}^n (1 - e^{-(q/V+\sigma)\Delta t})/(q/V + \sigma) \qquad 10$$

$$m_{out}^n = qM^n /V \qquad 11$$

and then along the piston flow pathway (P component) using

$$m_{spr}^n = m_{out}^{n-l} \exp(-\sigma l) \qquad 12$$

where M^n = annual average mass of nitrogen in the well-mixed reservoir in year *n* (g), m_{in}^n= annual nitrogen drainage into the reservoir in year *n* (g y^{-1}), m_{out}^n = annual nitrogen massflow leaving the reservoir in year *n* (g y^{-1}), m_{spr}^n = annual nitrogen massflow emerging as springflow in year *n* (g y^{-1}), σ = slowflow nitrogen attenuation coefficient (y^{-1}), *l* = piston flow lag time (y), *V* = long term average volume of water in the reservoir and *q* = long term average drainage (m^3 y^{-1}). Note that Eq. 10-12 apply in each sub-catchment *i* but the suffix *i* has been omitted. The mean residence time in groundwater of nitrogen from sub-catchment *i* is

$$MRT_i = V_i/q_i + l_i \qquad 13$$

Morgenstern et al. (2004, 2015) estimated MRT values for aquifers draining to major stream inflows to Lake Rotorua by fitting exponential piston flow models (EPM) to measured tritium concentrations in water samples collected from the streams during baseflow conditions (Table 2-1). Morgenstern modelled each aquifer as either one (Morgenstern et al. 2004) or two (Morgenstern et al. 2015) EPM combinations, whereas ROTAN-Annual modelled an EPM for each sub-catchment (the number of sub-catchment within each of Morgenstern's aquifers varied from 8 to 39).

ROTAN-Annual assumes that the mean residence time MRT_i for sub-catchment *i* varies with the distance between where drainage occurs and where springflow re-emerges.

$$MRT_i = \vartheta x_i \qquad 14$$

where x_i = Euclidean distance from the centroid of REC sub-catchment *i* to the spring (km) and ϑ = a factor inversely related to groundwater velocity and affected by sinuosity, which is assumed spatially uniform in each aquifer. The value of ϑ was calibrated so that the weighted average MRT for all sub-catchments draining to a sampling site matched published MRT values.

$$\vartheta = MRT \sum q_i / \sum q_i \, x_i \qquad 15$$

where i = sub-catchment number, summation is over all sub-catchments that drain to the site where *MRT* was measured, and q_i = long-term average drainage from sub-catchment i ($m^3 y^{-1}$). The residence time of the exponential flow component is

$$V_i/q_i = \epsilon_i MRT_i \qquad 16$$

and the piston flow lag is

$$l_i = (1 - \epsilon_i)MRT_i \qquad 17$$

where ϵ_i = proportion of exponential flow specified from data in Table 2-1. There is no mixing between the reservoirs or streamtubes associated with adjacent sub-catchments, but several streamtubes may re-emerge at the same spring where mixing occurs.

Table 2-1: Mean residence times (MRT) estimated by fitting binary exponential (E) piston flow (P) models (EPM) to tritium measurements in the major stream inflows to Lake Rotorua. ϵ is the proportion of exponential flow. Column 5 contains ϵ values for two models – the weighted average (brackets) is used in ROTAN-Annual. Source: Morgenstern et al. (2004, 2015).

Stream	Morgenstern et al. 2004		Morgenstern et al. 2015	
	MRT year	ϵ %	MRT year	ϵ %
Hamurana	110	100	125	82, 77 (80)
Awahou	61	100	75	100, 91 (99)
Waiteti	40	95	45	100, 90 (98)
Ngongotaha	15.5	80	30	100, 91 (98)
Waiowhiro	41.5	65	40	63 (63)
Utuhina	48	100	60	60, 100 (79)
Puarenga	37	100	40	100, 100 (100)
Waingaehe	127	100	145	94, 100 (95)
Waiohewa	41.5	95	40	100, 100 (100)

2.11 Quickflow attenuation

In each sub-catchment quickflow attenuation is assumed to depend on the distance of land from the stream. The length scale used is the minor axis of the ellipse whose area and perimeter are known from the REC. The minor axis B is estimated iteratively from

$$Area = \pi\, A\, B \qquad 18$$

$$Perimeter \sim 2\pi \sqrt{\frac{A^2+B^2}{2}} \qquad 19$$

where A = major axis, and B = minor axis. The massflow of nitrogen entering the stream via quickflow M_q (kg y^{-1}) is

$$M_q = p_q\, Loss\, \exp(-\kappa B) \qquad 20$$

where *Loss* = total annual nitrogen loss from OVERSEER (kg y^{-1}), p_q = proportion of quickflow and κ = quickflow attenuation coefficient (km^{-1}). κ was estimated by calibration to observed TN concentrations at ten stream monitoring sites. The value of κ was assumed spatially homogeneous across all sub-catchments draining to a particular stream monitoring site, but could vary between monitoring sites.

2.12 Streamflow attenuation

ROTAN-Annual assumes that nitrogen attenuation in streams is first-order with respect to distance.

$$N(x) = N(0)\exp(-\gamma x) \qquad 21$$

where x = distance downstream (km), N = nitrogen concentration (g m^{-3}) and γ = streamflow attenuation coefficient (km^{-1}). Viner (1987) showed that the streamflow attenuation coefficient decreased with increasing stream flow – reflecting the decrease in ratio of streambed area to volume:

$$\gamma = \alpha Q^{\beta} \qquad 22$$

where Q = stream flow (m^3 s^{-1}). A fixed β = -0.7 was assumed based on Viner (1987), and α was varied during calibration.

2.13 Nitrogen calibration

During calibration, trends in predicted nitrogen were compared visually with trends in observed DIN and this information used qualitatively when discussing the goodness of fit of the model. The formal assessment of model goodness of fit, however, was made using only stream TN concentrations. The fitting criterion was the root mean square percentage error (RMSE). At each monitoring site the RMSE was calculated from

$$\varepsilon = sign(\sum(prd - obs))\sqrt{\frac{\sum((prd-obs)/obs)^2}{n}} \qquad 23$$

where *prd* = annual average predicted nitrogen concentration (g m^{-3}) and *obs* = flow-weighted annual average observed TN concentration (g m^{-3}). Summation was over 3-4 years in each of eight time periods (1975-1977, 1987-1989, 1992-1995, 1996-1999, 2000-2003, 2004-2007, 2008-2011 and 2012-2015).

Calibration was performed using either ‘hill climbing’ or ‘grid searching’. The three key coefficients calibrated were the quickflow, slowflow and streamflow attenuation coefficients.

Using ‘hill climbing’, the quickflow coefficient was adjusted at each monitoring site holding the other two coefficients constant. Then the slowflow coefficient was adjusted and finally the streamflow coefficient was adjusted. The cycle was repeated until calibration was achieved – typically after 200-400 iterations. When ‘hill climbing’ was used:

$$\theta^{n+1} = \theta^{n} + \omega\frac{\varepsilon^{n}}{(\varepsilon^{n-1} - \varepsilon^{n})}(\theta^{n} - \theta^{n-1}) \qquad 24$$

where θ = coefficient value, ε = RMS error (Eq. 23), ω = under-relaxation factor and n = iteration number.

The ‘hill climbing’ method sometimes failed to converge and ‘grid searching’ was used:

$$\theta^{n+1} = \theta^{n} + \delta \quad \text{if } \varepsilon^{n} < 0 \qquad 25a$$

$\theta^{n+1} = \theta^n - \delta$ if $\varepsilon^n > 0$ 25b

where δ = user specified step size.

Calibration was undertaken assuming spatially homogeneous coefficients (viz., no difference between sub-catchments). After each iteration, the coefficients θ^{n+1} in Eqs 24 or 25 varied between monitoring sites but either: the weighted average of θ was calculated (where the weighting factors were the average stream flows at each site) (FWT), or the median value of θ was calculated (MED), and these values assigned across the entire catchment. Calibration was also undertaken allowing coefficients to vary between sub-catchments and streams within constraints based on published information (streamflow and slowflow attenuation), or assumed upper and lower bounds (quickflow attenuation).

Calibration was considered satisfactory when ε <10%. Lower values of ε could have been sought but 10% approximates the 95% uncertainty in long term average stream concentration. Thus calibration makes allowance for uncertainties in measured stream nitrogen.

2.14 Upper and lower bound attenuation coefficients

Constraints were imposed on the feasible range of coefficients using published values (streamflow attenuation) and an analysis of concentrations in 'old' groundwater (slowflow attenuation).

Table 2-2: Upper and lower bounds for model coefficients and 'most likely' values.

Coefficient	Symbol	Max	Mid	Min	Equation
Slowflow	σ	0.01	0.001	0.0001	4, 6
Quickflow	κ	10	0.1	0.001	12
Streamflow	α	0.45	0.095	0.01	14

Figure 2-3 shows published streamflow attenuation coefficients (Viner 1987). These data furnish likely bounds on coefficient α ($0.01 < \alpha < 0.5$ km^{-1}) and justify fixing β = -0.7 (see Eq. 22).

A literature search failed to locate information about quickflow attenuation in Rotorua soils and so a wide range was assumed (Table 2-2).

There are very limited data on groundwater concentrations from which make *a priori* estimates of slowflow attenuation, and so initially a wide range was assumed. Table 2-3 summarises the range of observed DIN concentrations in groundwater and in springs measured in July 2003 (Morgenstern et al. 2004). Also shown are estimates of drainage (taken as 50% of rainfall) and nitrogen loss. Drainage concentration and the slowflow attenuation coefficient are:

$$N_d = 10^5 \frac{L}{D} \qquad 26$$

$$\sigma = \ln\left(\frac{N_d}{N_o}\right)/MRT \qquad 27$$

where L = nitrogen loss rate in the sub-catchment (kg ha^{-1} y^{-1}), D = drainage (mm y^{-1}), N_d = estimated drainage concentration (mg m^{-3}), N_o = observed concentration in 'old' groundwater or spring flow (mg m^{-3}), MRT = groundwater age (y) and σ = slowflow attenuation coefficient (y^{-1}). Negative values of σ were omitted. From Table 2-3 independent estimates of the slowflow attenuation coefficient were made 0.0030 < σ < 0.0082 y^{-1} to refine (viz., reduce the uncertainty of) values estimated during calibration.

Table 2-3: Groundwater and spring data used to estimate the slowflow attenuation coefficient. Also shown is the range used to refine the calibration. N_o and *MRT* were from Morgenstern et al. (2004). *L* and *D* were from this study. σ is blank where Eq. 27 gave a negative value.

Site	N_o mg m^{-3}	L kg ha^{-1} y^{-1}	*MRT* y	D mm y^{-1}	N_d mg m^{-3}	σ y^{-1}
HAM	735	3	145	1100	273	
HAM	735	5	145	1100	455	
HAM	735	10	145	1100	909	0.0015
AWA	1410	5	64	1100	455	
AWA	1410	10	64	1100	909	
AWA	1410	15	64	1100	1364	
HATCHERY	945	5	54	1100	455	
HATCHERY	945	10	54	1100	909	
HATCHERY	945	15	54	1100	1364	0.0068
BARLOWS	567	5	73	1100	455	
BARLOWS	567	10	73	1100	909	0.0065
BARLOWS	567	15	73	1100	1364	0.0120
2116	223	3	135	1050	286	0.0018
2116	223	5	135	1050	476	0.0056
2116	223	10	135	1050	952	0.0108
1561	606	3	112	950	316	
1561	606	5	112	950	526	
1561	606	10	112	950	1053	0.0049
3691	847	3	94	1100	273	
3691	847	5	94	1100	455	
3691	847	10	94	1100	909	0.0008
					average	0.0056
					95%CI	0.0026

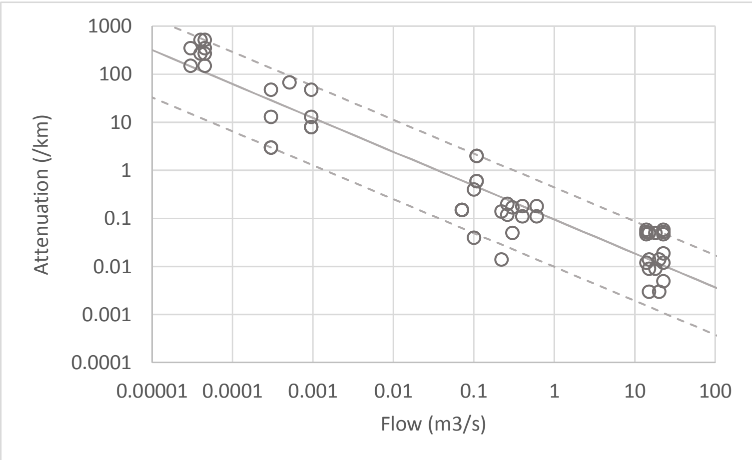


Figure 2-3: Reported streamflow attenuation coefficients. Source: Viner (1987). Dashed lines enclose 95% of observations. Lines: upper dashed α = 0.45, middle solid α = 0.095 and lower dashed α = 0.01. Slope σ = -0.705 for all lines.

3 Results

3.1 Stream flow

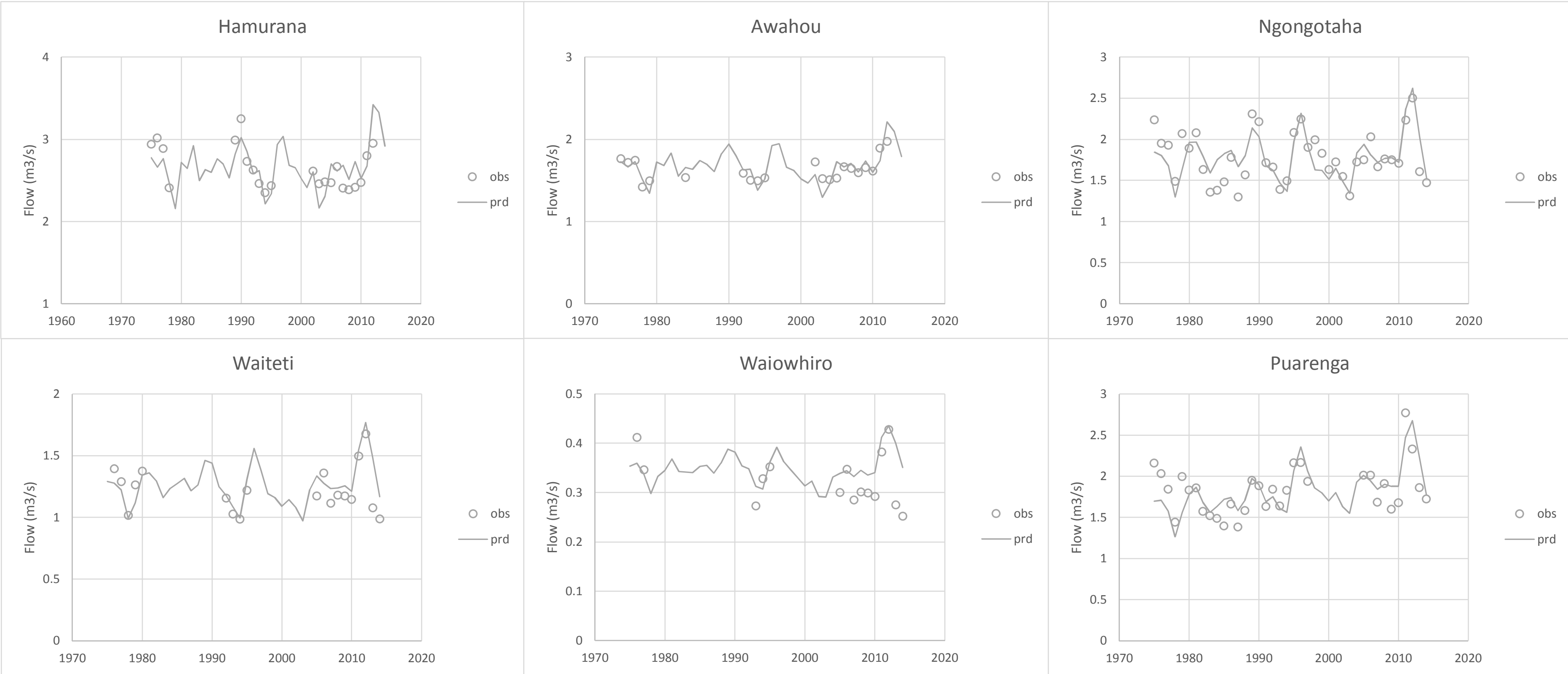
(
Table 3-1) summarises the calibrated drainage and groundwater sub-model coefficients. The mean difference (observed-predicted) nowhere lay outside the 95% confidence interval for the average observed flow (Table 3-2). The good fit for the Ohau Channel (the lake outlet) indicates that the water balance for the catchment as a whole was satisfactory after calibration of the plant-available water coefficient *w* (Eq. 1). Reservoir outflow coefficients (ρ in Eq. 2) of 0.9-1.5 y^{-1} gave a satisfactory match to observed year-to-year fluctuations in annual flow in most streams (Figure 3-1) but lower values (0.4-0.7 y^{-1}) were required where spring flow dominated.

Table 3-1: Summary of drainage fraction (*D*) and reservoir outflow coefficient (ρ). The drainage fraction was estimated by baseflow separation at the ten flow recorder sites. The outflow coefficient was estimated by visually matching observed and predicted year-to-year fluctuations in annual flow.

Parameter	Eq.	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
D	26	88%	92%	68%	63%	67%	62%	84%	79%	62%	74%
ρ	2, 3	0.60	0.50	1.50	1.00	0.90	1.00	0.60	1.00	0.40	0.90

Table 3-2: Observed minus predicted stream flows for the period 1975-2014 after calibration. Mean = average difference (observed-predicted) over the period 1975-2014 (L s^{-1}). 95%CI = 95% confidence interval for the observed annual flows (L s^{-1}).

Site	Mean	95%CI
Ohau Channel	34	405
Hamurana	-4	77
Awahou	0	50
Ngongotaha	11	91
Puarenga	-16	82
Utuhina	23	74
Waingaehe	8	3
Waiowhiro	-18	17
Waiohewa	-6	19
Waiteti	-24	59
Waipa	25	34


Hamurana
Awahou
Ngongotaha
Waiteti
Waiowhiro
Puarenga
Flow (m3/s)
obs
prd

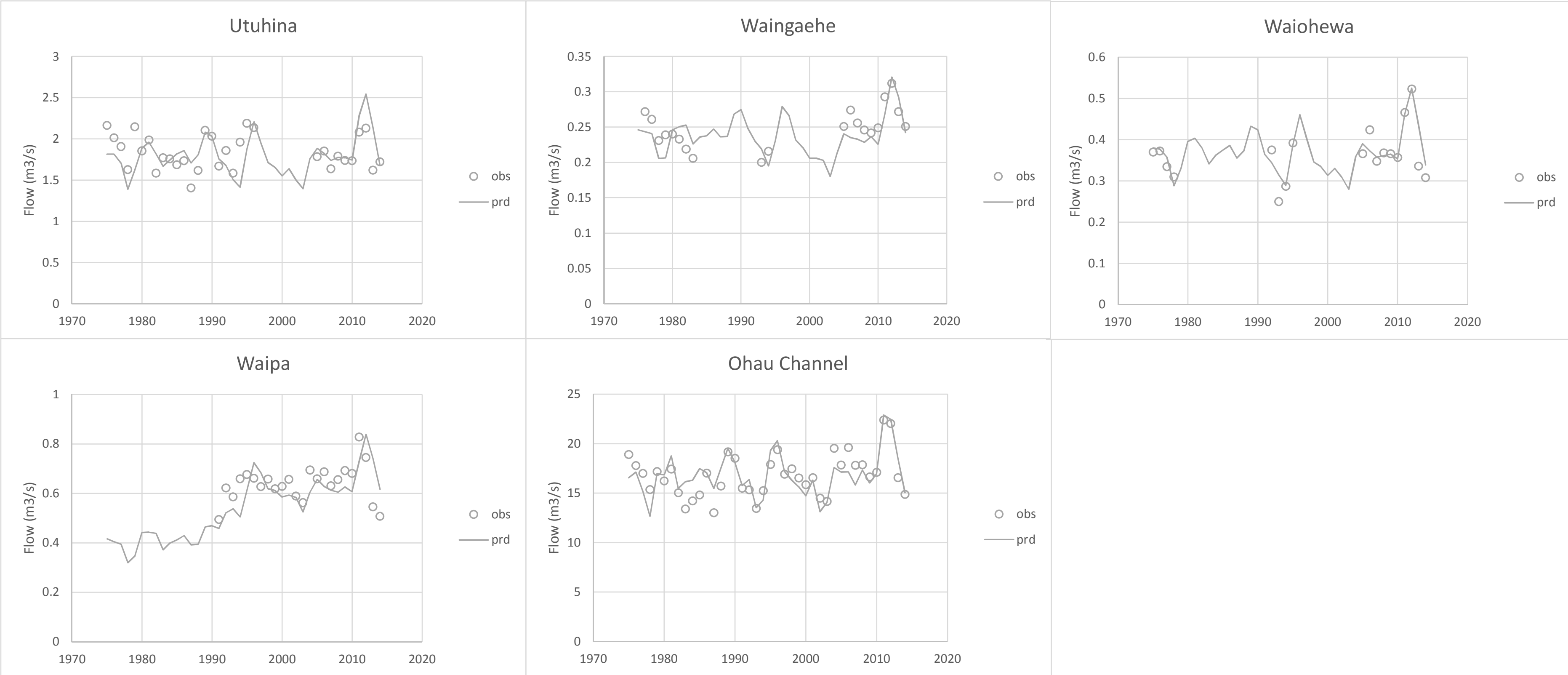


Figure 3-1: Observed versus predicted stream flows for the period 1975-2014 after calibration.

3.2 Predicted nitrogen concentrations – zero attenuation

Figure 3-2 compares observed annual average dissolved inorganic nitrogen (DIN) and total nitrogen (TN) concentrations, with model predictions made assuming no attenuation (viz., no nitrogen loss between forest or farmland and the lake). Groundwater travel times were those from Morgenstern et al. (2004).

In the Puarenga (PUA) and Waipa (WAI), the model captured the timing of increases in nitrogen concentrations coinciding with commissioning of the Rotorua Land Treatment Scheme (RLTS) in the early 1990s. However, predicted concentrations exceeded observations, implying high attenuation. In the Hamurana (HAM), Awahou (AWA) and Waiteti (WTT), observed nitrogen is predominantly in the form of DIN. The trend of increasing observed DIN concentrations was captured by the model. Attenuation is lower in the HAM than either the AWA or WTT.

In the Ngongotaha (NGO), predicted nitrogen concentrations increased monotonically with time, as does observed DIN, albeit at much lower concentrations. However, observed TN concentrations show a weak negative trend – a different pattern from DIN. Predicted concentrations in the Waiowhiro (WWH) decreased over time – dairy farming occurred in the catchment in the 1940-50s but ceased in the 1970s. In the Utuhina (UTU) the model captured the trend of decreasing observed DIN and TN. Attenuation is low in the UTU.

Nitrogen in the Waiohewa (WHE) is strongly influenced by geothermal inflows from Tikitere, and land use intensification appears not to have caused increases in either observed or predicted nitrogen concentration. Attenuation is high for DIN but low for TN.

High TN observations in the Waiowhiro (1993, 1994), Waingaehe (1994), Waiohewa (1993, 1994, 1995) and Utuhina (1993, 1994, 1995), together with low values in the Puarenga (1993) and Waingaehe (1993) appear to be outliers. This raises concerns about the small number, and reliability, of TN observations during the 1990s, and these observations were not used during model calibration.

There are differences between streams in the time trends of nitrogen concentration. Six streams show a steady increase in observed DIN (PUA, WAI AWA, HAM, WNG and WTT) while four show little overall change (UTU, WHE, NGO, WWH). The NGO shows an increase from 1970-2000 and a decrease thereafter. The trends in observed TN are not as noticeable as trends in DIN partly because few TN observations exist prior to the 1990s.

Table 3-3 indicates low attenuation in the Hamurana (RMSE 14%), but high attenuation in the Ngongotaha (85%) and Puarenga (83%).

Table 3-3: Root mean square difference (RMSE) between predicted unattenuated nitrogen and observed TN concentrations as a percentage of observed concentration. Positive values indicate that predictions exceed observations. The RMSE is an indicator of the amount of attenuation. AWA = Awahou. HAM = Hamurana. NGO = Ngongotaha. PUA = Puarenga. UTU = Utuhina. WHE = Waiohewa. WNG = Waingaehe. WTT = Waiteti. WWH = Waiowhiro. WAI = Waipa.

Catchment	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
RMSE difference (predicted vs. observed)	42%	14%	85%	83%	29%	24%	20%	58%	29%	52%

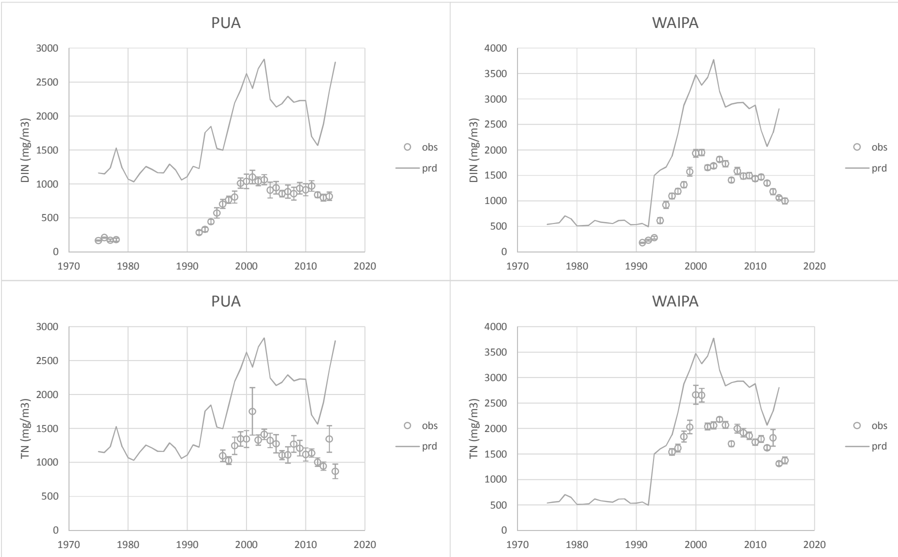


Figure 3-2: Comparison of observed DIN and TN concentration (circles) and predicted unattenuated nitrogen concentration (lines) in the major streams. (Continued overleaf).

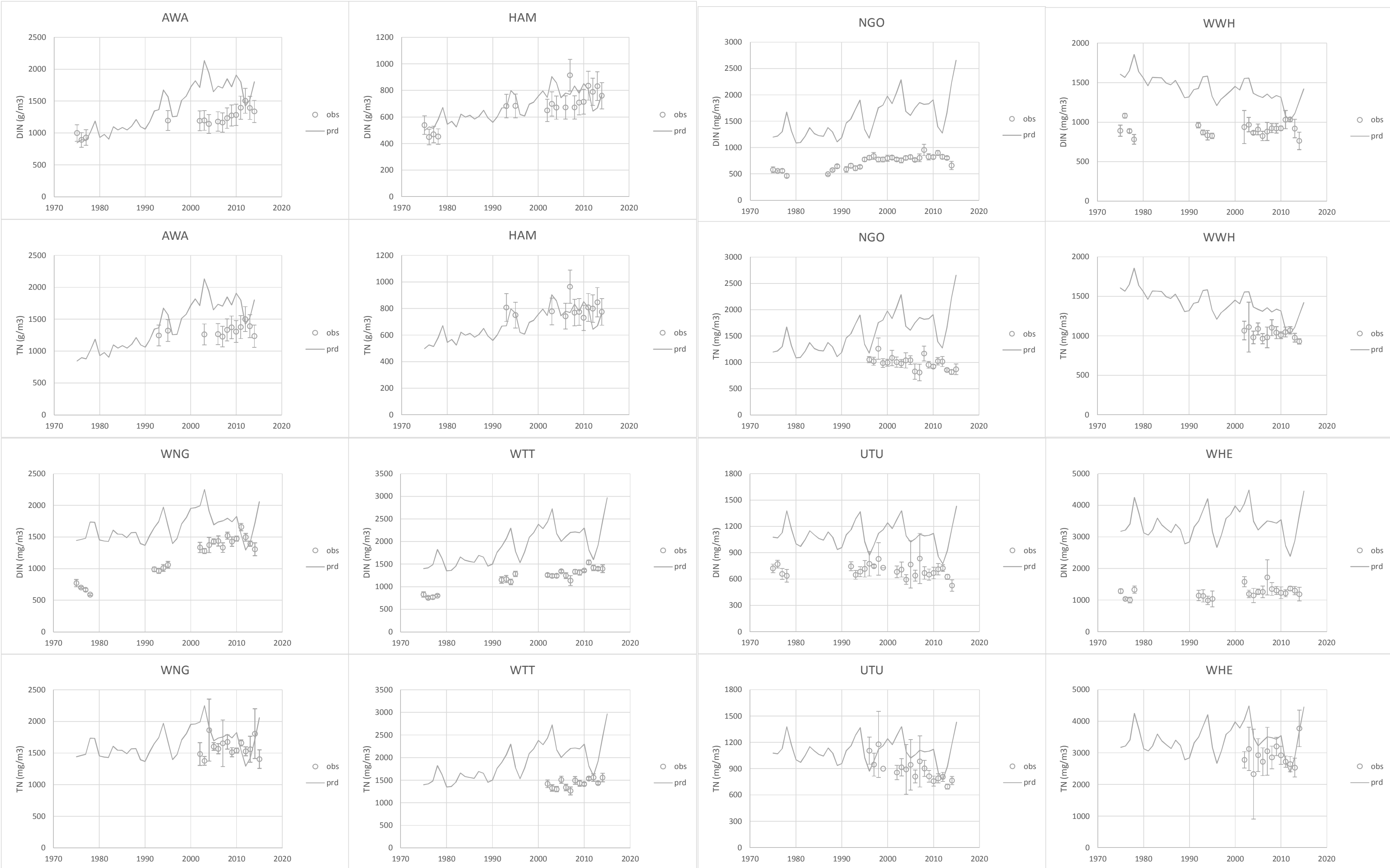


Figure 3-2: Comparison of observed DIN and TN concentration (circles) and predicted unattenuated nitrogen concentration (lines) in the major streams. Error bars are the 95% confidence interval of flow-weighted average observed concentrations. Predictions assume zero attenuation which is unrealistic, but enables an initial assessment of attenuation and time trends.

3.3 Calibration 1: Spatially homogeneous attenuation

Table 3-4 and Table 3-5 summarise model coefficients and RMSE values where coefficients were assumed to be spatially homogeneous and calibrated to minimise either the flow-weighted average (FWT) or the median (MED) of the RMSE values at the ten monitoring sites. The starting values of the three attenuation coefficients calibrated were their mid-range values (see Table 2-2), and groundwater travel times were those from Morgenstern et al. (2004).

Calibrations were satisfactory ($\varepsilon < \pm10\%$) at only two (FWT – HAM and WAI) and three (MED – HAM, PUA and WWH) sites. The model significantly over-estimated TN concentrations ($\varepsilon > 15\%$) in the AWA and significantly under-estimated concentrations ($\varepsilon < -15\%$) in the UTU, WHE and WNG. The FWT calibration furnished higher attenuation coefficients than the MED calibration.

Using the ROTAN-2011 model, Rutherford et al. (2011) predicted a steady-state input to the lake of 725 t y^{-1} (excluding rainfall falling on the lake of 30 t y^{-1}) if current land use remained unchanged. When calibrated assuming spatially homogeneous attenuation coefficients, ROTAN-Annual predicted steady-state load 8% (FWT) and 20% (MED) higher than ROTAN-2011.

Table 3-4: Calibration 1A. Coefficients were assumed to be spatially homogeneous and set equal to the flow-weighted mean (FWT) across monitoring sites. Positive RSME indicates that the model over-estimates observed concentrations. Note [a] excluding nitrogen in rain falling directly on the lake.

Parameter	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.032									
1000σ	3.2									
α	0.089									
RMSE	54%	5%	13%	-14%	-25%	-34%	-36%	-15%	-17%	-7%

	Nitrogen inflow at steady-state assuming current land use				
FWT	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	Difference %
	638	146	784 [a]	725 [a]	+8%

Table 3-5: Calibration 1B. Coefficients were assumed to be spatially homogeneous and set equal to the median (MED) across monitoring sites.

Parameter	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.025									
1000σ	2.5									
α	0.050									
RMSE	60%	9%	19%	9%	-19%	-23%	-23%	-11%	-9%	13%

	Nitrogen inflow at steady-state assuming current land use				
MED	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 2011 t y^{-1}	Difference %
	722	146	916	725	+20%

Figure 3-3 compares observed and predicted nitrogen concentrations after Calibration 1A (FWT). In the Awahou (AWA) and Hamurana (HAM), total nitrogen concentration is predominantly in the form of nitrate, TN and DIN concentrations are similar, and the model captures the increasing concentration trends with time. Model fit is acceptable in the HAM, but the model significantly over-estimates DIN and TN concentrations in the AWA.

The model fits observed TN in the Ngongotaha (NGO) and captures the increasing trends over time in observed DIN, albeit at higher concentrations. In the Waiowhiro (WWH) and Utuhina (UTU) the model under-estimates TN. Observed DIN in the Waiohewa (WHE) makes up only 25-30% of TN – the balance is organic and particulate nitrogen associated with the growth of nitrifying micro-organisms in the stream bed (Williamson & Cooke 1982). There are no obvious time trends in either DIN or TN in the WHE – the large and steady geothermal inflow from Tikitere appears to swamp any effects of land use change. The model under-estimates observed TN. The model captures the pattern of increasing nitrogen concentration in the Waiteti (WTT), where nitrate is the predominant form of nitrogen, but under-estimates recent TN concentrations. In the Waingaehe (WNG), the model predicts little change over the period 1975-2015, whereas observed DIN increased significantly – the reasons for this difference are unclear. The model consistently under-estimates observed TN concentrations. In the Waipa (WAI) and the Puarenga (PUA) (into which the Waipa flows), observed and predicted nitrogen concentrations both increase significantly in the early 1990s, reflecting the influence of the RLTS. The model matches observed TN concentrations at both sites.

For Calibration 1B (using median rather than flow-weighted averages), the general patterns shown in Figure 3-3 were similar (details omitted for brevity).

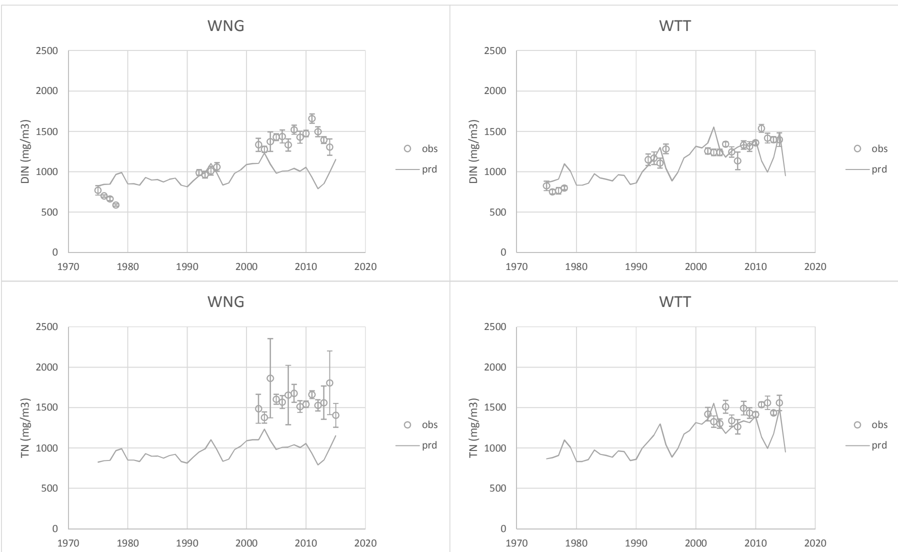


Figure 3-3: Calibration 1A results. (Continued overleaf).

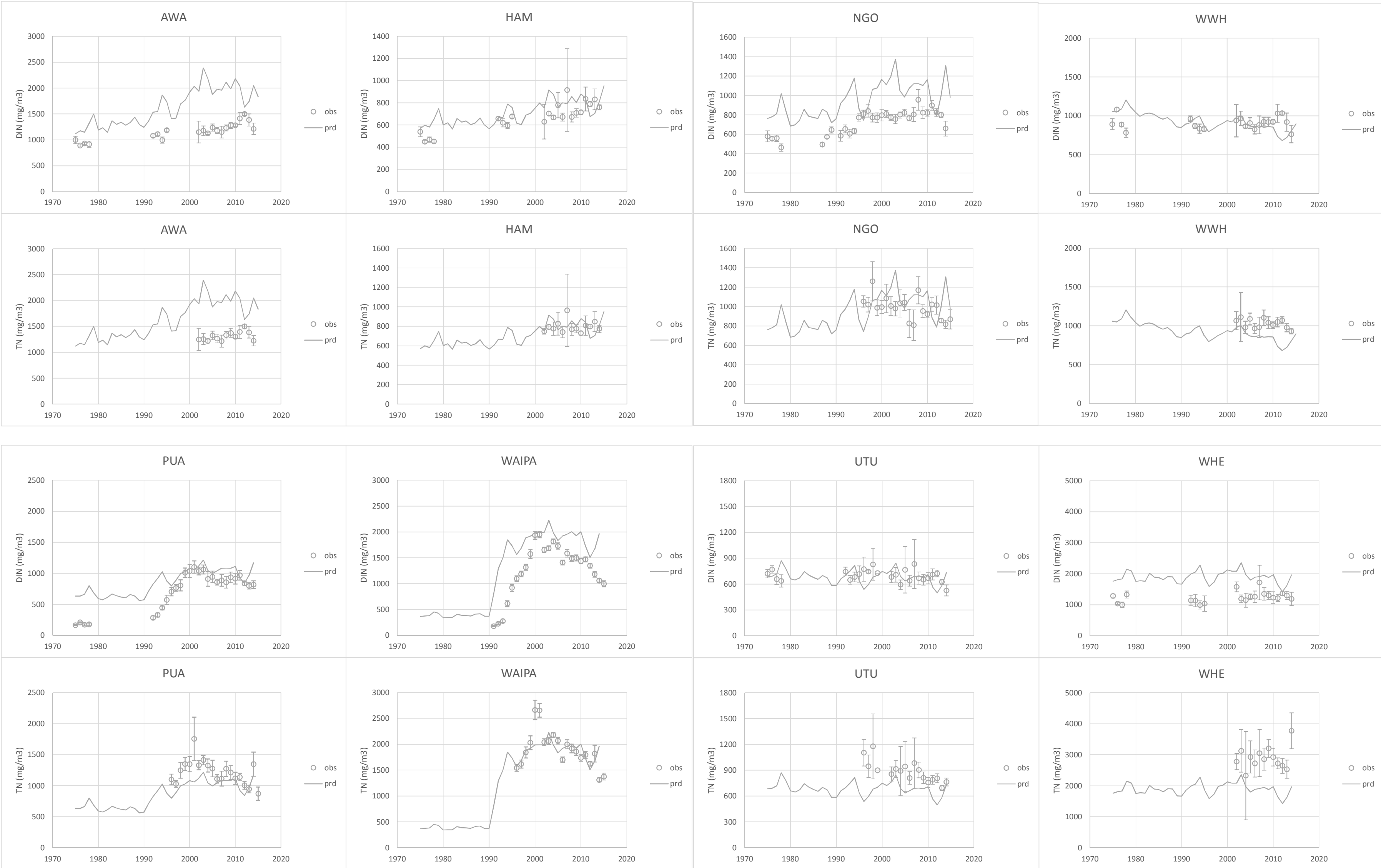


Figure 3-3: Calibration 1A results. Observed (circles) and predicted (lines) DIN and TN concentration in the major streams. Error bars are the 95% confidence interval of flow-weighted annual average concentration. Coefficients were assumed spatially homogeneous and were set to the flow-weighted average of values at the 10 monitoring sites.

3.4 Calibration 2: Uniqueness of attenuation calibration

Table 3-6 summarises results of repeating Calibration 1 starting with different initial values of the three attenuation coefficients – the eight possible combinations of the *a priori* upper and lower bound values.

Each calibration gave a similar flow-weighted average RMSE value ε across the ten monitoring sites. However, attenuation coefficients differed significantly between calibrations. The most extreme differences occurred for quickflow attenuation, where $0.001 < \kappa < 6.31$ km^{-1}. In four sub-catchments (AWA, HAM, WNG and WTT) drainage was high, quickflow was only 8-21% and predicted stream nitrogen concentrations were insensitive to quickflow attenuation. In the other six sub-catchments quickflow was 26-38% and quickflow attenuation had a detectable effect on predicted concentrations.

Calibrated slowflow attenuation coefficients σ were always below the *a priori* maximum (0.1 y^{-1}). On two occasions, however, σ remained at the *a priori* minimum (0.001 y^{-1}), when quickflow and/or streamflow attenuation accounted for the majority of nitrogen reduction. This suggests that the minimum slowflow attenuation coefficient may be less than 0.001 y^{-1}. Calibrated streamflow attenuation coefficients ranged from $0.011 < \alpha < 0.126$ km^{-1} within the *a priori* range of $0.01 < \alpha < 0.45$ km^{-1}.

Table 3-6: Calibration 2 results. Values of the three attenuation coefficients calibrated with eight different combinations of starting values. Nitrogen losses were the mid-range estimates and coefficients were assumed spatially homogeneous. RMSE is the flow-weighted value across the ten monitoring sites. Also shown are the predicted steady-state loads to the lake assuming current land use (SS) and the a priori maximum and minimum values used to constrain calibration.

Calibrated values			Starting values				SS
κ	σ	α	κ	σ	α	RMSE	t y^{-1}
1.26	0.036	0.011	10	0.1	0.01	1%	741
6.31	0.004	0.036	10	0.001	0.01	1%	793
0.158	0.001	0.126	10	0.001	1	1%	807
0.032	0.006	0.056	10	0.1	1	0%	775
0.001	0.001	0.126	0.001	0.001	1	1%	808
0.001	0.022	0.010	0.001	0.1	0.01	1%	700
0.001	0.006	0.056	0.001	0.1	1	0%	776
0.025	0.006	0.056	0.001	0.001	0.01	0%	776

Bound	κ	σ	α
Max	10	0.1	1
Min	0.001	0.001	0.01

Predicted steady-state loads to the lake assuming current land use varied from 700-808 t y^{-1} compared with 784 t y^{-1} estimated by Calibration 1 (which started with *a priori* mid-range coefficients) and 725 t y^{-1} estimated using ROTAN-2011. These estimates do not differ by more than ±7%.

3.5 Calibration 3: Spatially variable attenuation

Table 3-7 summarises the results of calibration allowing coefficients to differ between sub-catchments subject to the constraints in Table 2-2. High, middle and low nitrogen loss estimates were specified, the starting coefficients were the mid-range values, and groundwater travel times were those from Morgenstern et al. (2004).

Convergence was slow (viz., during calibration, coefficients changed at each step with very little effect on RMSE value) which is a further indication that different combinations of coefficients gave a similar goodness of fit. As discussed in the previous section, choosing different starting values is likely to have furnished different calibrated coefficients and the coefficient values in Figure 3-7 are not 'unique' or 'optimal'.

Using the lower bound loss estimates, Calibration 3A reached the *a priori* lower bound attenuation coefficients, and failed to converge (ε > 5%), at three sites. It also furnished lower bound coefficients for streamflow attenuation at six sites. These findings suggest that the lower bound losses under-estimate 'true' losses and hence that Calibration 3A coefficients should be regarded with caution.

One feature of Table 3-7 is the variation of attenuation coefficients between monitoring sites. In Calibrations 3B and 3C, quickflow and slowflow attenuation coefficients varied by a factor of 2-5, while streamflow attenuation coefficients varied by a factor of 40-50. Note that the choice of different starting values may alter the magnitude of such variations, but is unlikely to eliminate them.

Despite the variations in calibrated attenuation coefficients, allowing coefficients to vary spatially resulted in only small differences in the predicted steady-state lake loads (656-780 t y^{-1}) to the lake, assuming current land use, relative to those obtained with Calibration 2 (700-808 t y^{-1}).

Table 3-7: Calibration 3 results. Model coefficients calibrated assuming constrained spatial non-homogeneity. Losses are the low (top), mid (middle) and high (bottom) estimates. Cells coloured grey indicate where calibrated coefficients reached the *a priori* upper and lower bounds and/or calibration failed to converge. ROTAN-2011 is the estimate of steady-state lake load in Rutherford et al. (2011).

Calibration 3A – low loss estimates

	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.102	0.001	0.078	0.079	0.001	0.004	0.001	0.065	0.021	0.098
1000σ	11.0	0.01	8.10	7.94	0.01	0.40	0.01	6.48	2.09	9.81
α	0.16	0.01	0.04	0.03	0.01	0.01	0.01	0.01	0.01	0.09
RMSE	5%	-6%	0%	0%	-14%	0%	-15%	0%	0%	0%

Nitrogen inflow at steady-state assuming current land use

VAR	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	difference %
	651	128	780	725	+8%

Calibration 3B – mid loss estimates

	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.133	0.051	0.094	0.084	0.062	0.027	0.027	0.075	0.064	0.094
1000σ	13.7	5.1	9.5	8.4	6.2	2.8	2.7	7.5	6.4	9.5
α	0.47	0.01	0.08	0.04	0.01	0.01	0.01	0.02	0.01	0.08
RMSE	0.3%	0.0%	0.0%	0.1%	0.0%	0.0%	0.1%	0.2%	0.0%	0.0%

Nitrogen inflow at steady-state assuming current land use

VAR	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	difference %
	591	146	736	725	+2%

Calibration 3C – high loss estimates

	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.153	0.112	0.119	0.094	0.075	0.073	0.073	0.097	0.079	0.099
1000σ	15.3	11.2	12.5	9.5	7.5	7.3	7.3	9.6	7.9	9.9
α	0.84	0.07	0.30	0.08	0.02	0.02	0.02	0.08	0.03	0.10
RMSE	-0.3%	0.0%	0.1%	-0.3%	0.1%	0.0%	0.2%	0.1%	0.1%	0.0%

Nitrogen inflow at steady-state assuming current land use

VAR	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	difference %
	495	161	656	725	-10%

3.6 Calibration 4: Variation in groundwater travel time

Table 3-8 compares the results of calibration using two published estimates of groundwater travel times (Morgenstern et al. 2004 and 2015) (see Table 2-1).

Calibrated attenuation coefficients were identical in WAI (where MRT was identical), and similar in PUA, WWH and WHE (where MRT changed only slightly). Higher MRT values in the AWA, HAM, NGO and UTU resulted in lower slowflow attenuation. The higher MRT from Morgenstern et al. (2015) resulted in a steady-state load to the lake (766 t y^{-1}), only 4% larger than obtained using the lower MRT in Morgenstern et al. (2004) (736 t y^{-1}).

Table 3-8: Calibration 4 results. Calibrated model coefficients assuming two different estimates of groundwater travel time. Mid-range nitrogen losses and starting values were used. ROTAN-2011 is the estimate of steady-state lake load in Rutherford et al. (2011).

Morgenstern et al. (2004)

	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.133	0.051	0.094	0.084	0.062	0.027	0.027	0.075	0.064	0.094
1000σ	13.7	5.1	9.5	8.4	6.2	2.8	2.7	7.5	6.4	9.5
α	0.47	0.01	0.08	0.04	0.01	0.01	0.01	0.02	0.01	0.08
RMSE	0.3%	0.0%	0.0%	0.1%	0.0%	0.0%	0.1%	0.2%	0.0%	0.0%

Nitrogen inflow at steady-state assuming current land use

	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	Difference %
VAR	591	146	736	725	+2%

Morgenstern et al. (2015)

	AWA	HAM	NGO	PUA	UTU	WHE	WNG	WTT	WWH	WAI
κ	0.127	0.032	0.082	0.084	0.044	0.027	0.029	0.072	0.068	0.094
1000σ	13.1	3.2	8.2	8.4	4.4	2.7	2.9	7.2	6.9	9.5
α	0.30	0.01	0.04	0.04	0.01	0.01	0.01	0.02	0.01	0.08
RMSE	0.1%	0.0%	0.0%	-0.1%	0.0%	0.0%	0.1%	0.0%	0.1%	0.0%

Nitrogen inflow at steady-state assuming current land use

	Streams t y^{-1}	GW direct t y^{-1}	Total t y^{-1}	ROTAN-2011 t y^{-1}	Difference %
VAR	621	146	766	725	+6%

3.7 Calibration 5: Distribution of coefficients

The previous analysis indicates that no single combination of attenuation coefficients provided a satisfactory match between observed and predicted stream nitrogen concentrations at all ten monitoring sites. Rather, different combinations gave a satisfactory match depending on the model inputs (viz., nitrogen losses and groundwater travel times).

A technique commonly used to assess the sensitivity of calibration to uncertainties in model inputs is to undertake a 'Monte Carlo' analysis. If applied to ROTAN-Annual, this would involve selecting a large number of different combinations of nitrogen losses, groundwater travel times and initial coefficient values, and calibrating the model for each combination. Then the statistical distribution of the calibrated coefficients and the steady-state lake load could be determined.

There is a problem applying this approach: namely that the statistical distributions of the uncertainty of the inputs need to be known *a priori* (viz., the type of statistical distribution (normal, log-normal, uniform etc.,) the mean, maximum, minimum etc., and the correlations between inputs). Such information is not known for the inputs to ROTAN-Annual. The approach taken in this study was to undertake a sensitivity analysis as follows:

1. The quickflow attenuation coefficient was constrained to lie in the range $0.005 < \kappa < 1$ km^{-1}. As noted above, predicted concentrations were insensitive to quickflow attenuation because the majority of water and nitrogen drains to groundwater.
2. The slowflow attenuation coefficient was constrained to lie in the range $0.0001 < \sigma < 0.1$ y^{-1}.
3. The streamflow attenuation coefficient was constrained to lie in the range $0.01 < \alpha < 0.45$ km^{-1}.
4. Groundwater travel times were fixed using values derived from Morgenstern et al. (2004) because variations in travel time had only a minor effect on predicted lake loads.
5. Five estimates of nitrogen losses were specified: the most likely losses, two estimates between the minimum and most likely losses, and two estimates between the maximum and most likely losses. The uncertainty in losses was approximately normal.
6. The model was calibrated 1000 times using different combinations of inputs.

The quickflow attenuation coefficient was found to be uncorrelated with the slowflow and streamflow attenuation coefficients. The reason is that predicted stream nitrogen concentration is insensitive to quickflow attenuation because, in the model, only a small fraction of nitrogen loss travels via the quickflow pathway. As a result steady-state lake load, assuming current land use, was uncorrelated with quickflow attenuation (Figure 3-4).

There was, however, a significant negative correlation between the slowflow and streamflow attenuation coefficients (Figure 3-5). As slowflow attenuation increased, less streamflow attenuation was required in the model to match observed and predicted stream nitrogen concentrations. Steady-state lake load, assuming current land use, decreased as the slowflow and streamflow attenuation coefficients increased (Figure 3-6, Figure 3-7).

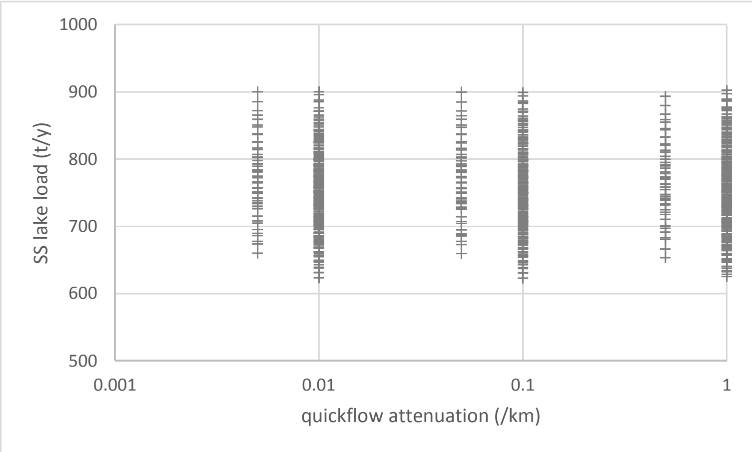


Figure 3-4: Correlations between predicted steady-state lake load assuming current land use and calibrated quickflow attenuation coefficient. Results presented for five estimates of nitrogen loss.

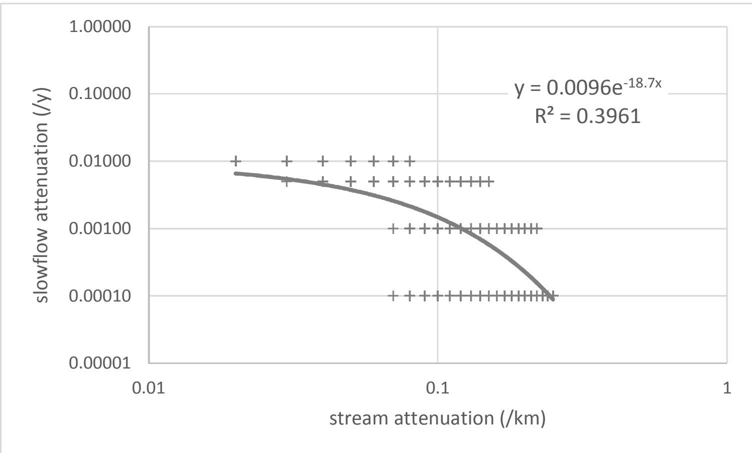


Figure 3-5: Correlation between calibrated slowflow and streamflow attenuation coefficients for three estimates of nitrogen loss (high, low and mid).

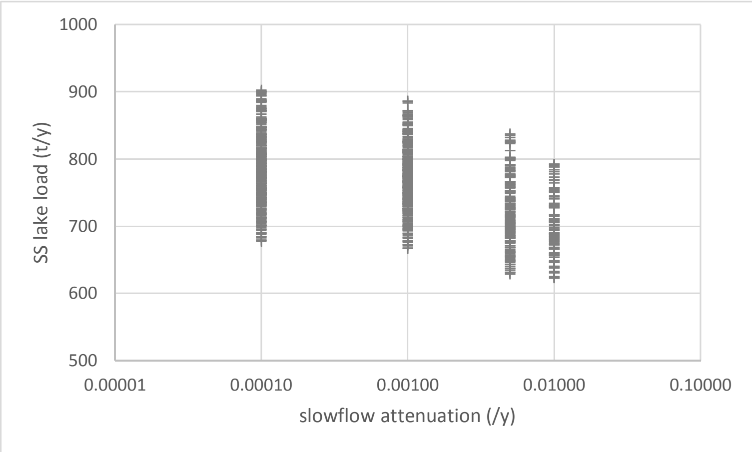


Figure 3-6: Correlation between calibrated slowflow attenuation coefficients and predicted steady-state lake loads assuming current land use. Results presented for five estimates of nitrogen loss.

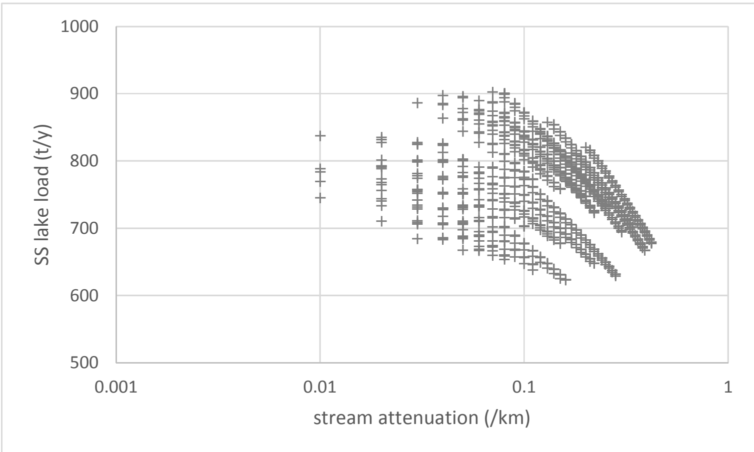


Figure 3-7: Correlation between calibrated streamflow attenuation coefficients and predicted steady-state lake loads assuming current land use. Results presented for five estimates of nitrogen loss.

3.8 Steady-state loads assuming current land use

Approximate frequency distributions of the steady-state lake load assuming current land use were calculated using the 1000 combinations of attenuation coefficients generated by Calibration 5. Assuming nitrogen losses remain at current levels and that the slowflow attenuation coefficient lies in the range $0.0001 < \alpha < 0.01$ y^{-1} (Table 2-2), then the 'most likely' steady-state lake load assuming current land use is 760 t y^{-1} with 95% confidence limits of 660-860 t y^{-1} (Figure 3-8). However, the synoptic sampling of Morgenstern et al. (2004) indicates slowflow attenuation coefficients in the range $0.0030 < \alpha < 0.0082$ y^{-1} (Table 2-3). Using these value the 'most likely' steady-state lake load assuming current land use is 750 t y^{-1} with 95% confidence limits of $670 < SS < 840$ t y^{-1}.

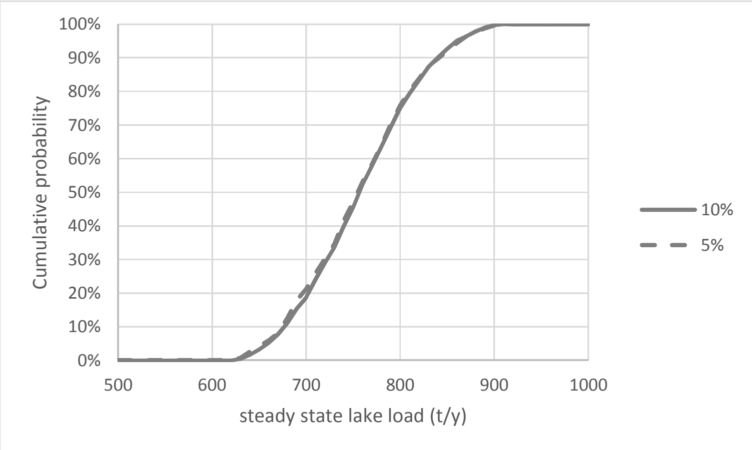


Figure 3-8: Approximate frequency distribution of steady-state lake load assuming current land use. Distributions are shown for calibrations with flow weighted mean RMSE values <5% and <10%.

3.9 Steady-state loads assuming reduced losses

Table 3-9 summarises the nitrogen loss reductions specified by BoPRC and their implementation dates. Reductions through 'Rules' and 'Incentives' were originally determined using OVERSEER v5 and have been scaled by 188% - the average increase for Dairy and Drystock using OVERSEER v6 (Alastair MacCormick, pers. comm.). Reductions for the three Engineering actions were not scaled. It was assumed that the spray disposal of wastewater in Whakarewarewa Forest (RLTS) ceased in 2020 although in the model nitrogen leaching from the site took over a decade to reach a new, lower steady state. The WWTP was assumed to discharge 30 t y^{-1} to the lake from 2020.

Assuming these reductions and slowflow attenuation coefficient in the range $0.0001 < \alpha < 0.01$ y^{-1}, the 'most likely' steady state lake load is 440 t y^{-1} with a 95% confidence interval of 390-490 t y^{-1} (Figure 3-9). Using the slowflow attenuation coefficients in the range $0.003 < \alpha < 0.082$ t y^{-1} alters the 'most likely' steady state lake load to 420 t y^{-1} with a 95% confidence interval of $390 < SS < 460$ t y^{-1}.

Table 3-9: Nitrogen loss reductions. Reductions through 'Rules' and 'Incentives' are scaled to account for differences between OVERSEER v5 and v6.

Mechanism	Agreed reduction t y^{-1}	Scaled reduction t y^{-1}	Implementation date
Land use			
Rules	140	263	Staged (2020, 2027, 2033)
Incentives	100	188	Staged (2027, 2033)
Gorse control	30	30	2033
Engineering			
Sewage reticulation	10	10	2020
Tikitere N removal	20-25	22.5	2020
Urban actions	15-20	17.5	Staged (2027, 2033)
Waste water treatment plant			
RLTS	56	56	Gradual reduction
WWTP	-30	-30	2020
TOTAL	**340**	**531**	

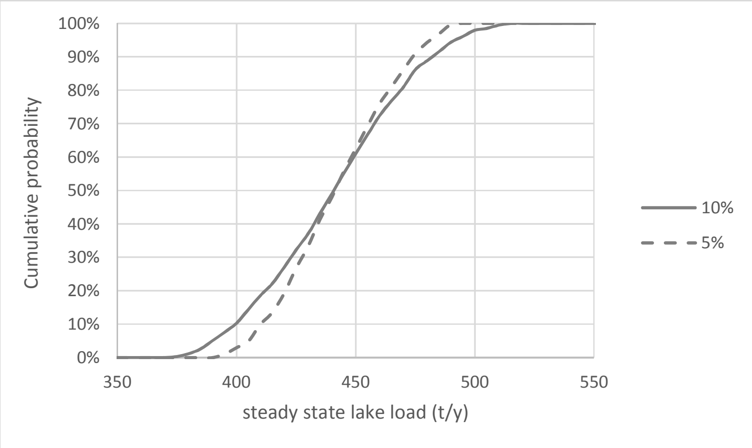


Figure 3-9: Approximate frequency distribution of steady-state lake load assuming reduced losses. Distributions are shown for calibrations with flow weighted mean RMSE values <5% and <10%.

3.10 Time scale of recovery

Figure 3-10 shows median predicted annual lake loads (excluding rainfall on the lake) for the period 1900-2100 assuming current nitrogen losses from 2015-2100. Upper (UCL) and lower (LCL) bounds are also shown. The steady increase in lake load from 1900-1980 was the result of land use intensification and population increase. The sudden decrease in the 1990s was the result of diverting wastewater to Whakarewarewa Forest although nitrogen losses from the RLTS gradually increased over the next decade. The steady-state load for these simulations (690 < *SS* < 840 t y^{-1}) was not reached by 2100 (viz., after 85 years) because of the very high MRT in some aquifers.

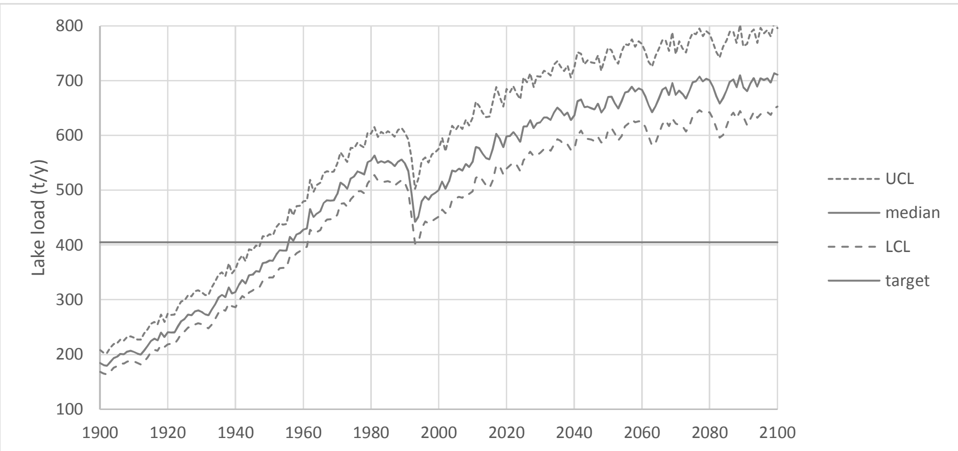


Figure 3-10: Predicted lake loads 1900-2100. From 2015-2100 losses are assumed constant at 2015 estimates. Results show the 'most likely' (median) loads together with approximate upper (UCL) and lower (LCL) bounds. The 'target' of 405 t y^{-1} excludes rain falling directly on the lake.

The nitrogen controls summarised in Table 3-9 were predicted to decrease lake loads by 110 and 140 t y^{-1} by 2043 (10 years after the final stage of loss reductions) and 2058 (+25 years) respectively (Figure 3-11). These are 60% and 70% of the reductions required to reach the target lake load. The rate of decrease was slow after c. 2050 such that by 2100 the annual lake load was still c. 20 t y^{-1} (10%) higher than the steady state load – the reason is that some aquifers (e.g., draining to the Hamurana, Awahou and Waingaehe streams) have very long residence times and will take several decades to reach steady state.

Rutherford et al. (2011), using ROTAN-2011, predicted that 10 years after loss reductions, lake loads occasionally dropped below the steady-state load, that after 25 years they were consistently within ±10-15% of the steady-state load, but that by 2100 they were still not quite at the steady-state. This is a faster rate of recovery than predicted by ROTAN-Annual.

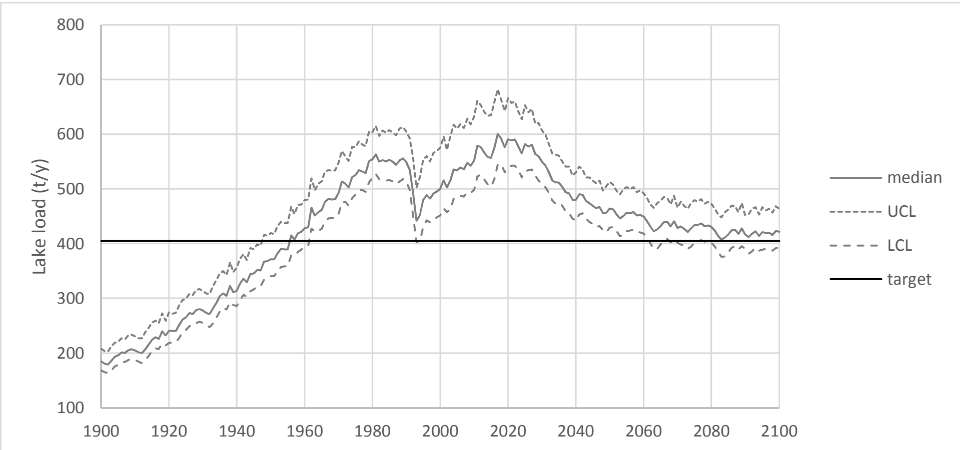


Figure 3-11: Predicted lake loads 1900-2100. From 2010 staged reductions in nitrogen loss are modelled (see Table 3-9). Results show the 'most likely' (median) loads together with approximate upper (UCL) and lower (LCL) bounds. The 'target' of 405 t y^{-1} excludes rain falling directly on the lake.

3.11 Catchment-scale attenuation

For the 1000 simulations of Calibration 5 average attenuation across the whole catchment was calculated by comparing the steady-state lake load assuming current land use with the current losses.

The 'most likely' value of catchment-scale is 42% with a 95% confidence interval of 32%-50% (Figure 3-12). Catchment-scale attenuation at Rotorua estimated in this study are in accord with published values that are typically of the order 50% (e.g., Alexander et al. 2002).

Rutherford et al. (2011) used ROTAN-2011 and OVERSEER v5 found that observed and predicted stream nitrogen concentrations matched assuming zero attenuation in all but the Puarenga catchment. They noted this level of attenuation was unexpectedly low. In this study OVERSEER v6 was used to estimate nitrogen losses which on average were 88% higher than losses estimated using OVERSEER v5 (Alastair MacCormick, pers. comm.) Using the higher OVERSEER v6 losses brings the catchment-scale attenuation into line with published values.

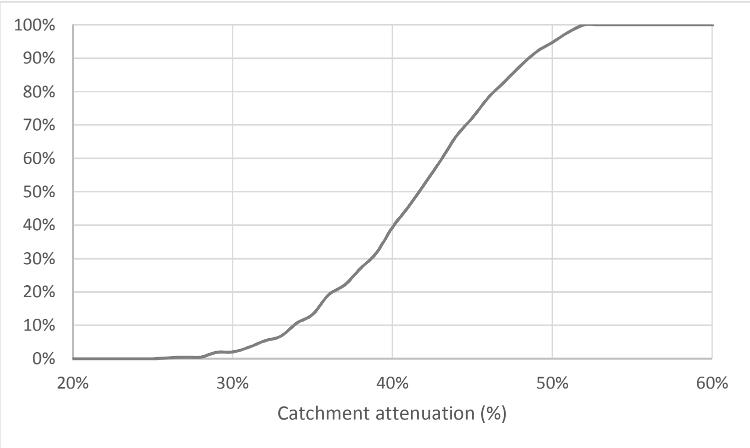


Figure 3-12: Frequency distribution of catchment-scale attenuation.

4 Discussion

Observed and predicted annual flows matched within the 95% confidence interval of observed flows. The water balance included groundwater (but not streamflow) from an 'extra' area outside the surface catchment defined by White et al. (2014). The coefficient *w* in the Zhang model (Eq. 1) was adjusted to match the average lake outflow and hence calibration compensates for any errors in the 'extra' area, the spatial distribution of rainfall, measured stream flows and estimated PET.

In some parts of the watershed the aquifer and surface boundaries coincided. Where this occurred, varying where groundwater re-emerged along the stream channel had no effect on predicted flow at the recorder site. In other parts of the watershed the only way to match observed and predicted stream flow (e.g., in the Hamurana) was to assume that groundwater crossed surface sub-catchment boundaries. Where surface catchment and aquifer boundaries did not coincide, the percentage drainage and the point where groundwater emerged had a significant effect on predicted stream flow. There is very little data about the longitudinal variation of flow in Rotorua streams even though the location of major springs is known. Consequently there is uncertainty about where groundwaters re-emerge in some of the stream channels. As a result, in some catchments (e.g., the Ngongotaha, Utuhina and Puarenga), it is difficult to separate the effects of slowflow (viz., groundwater) attenuation and streamflow attenuation. Additional information about flow distributions along stream channels would improve calibration of flows and attenuation in groundwater and streams.

ROTAN-Annual routes nitrogen through the catchment using mass flux (viz., $t\ y^{-1}$) which is consistent with OVERSEER which estimates annual nitrogen losses. Attenuation coefficients were calibrated using annual average stream concentrations (flux/flow) and hence predicted flows would have affected calibration. However, errors in flow predictions were small (especially when compared with uncertainties in nitrogen loss, groundwater flow paths and residence times), and are unlikely to have a significant effect on model calibration.

In some catchments the time trends of observed and predicted nitrogen matched. In other catchments, however, trends differed. Possible explanations include errors in: land use history, nitrogen loss estimate, groundwater flow pathways and groundwater travel times.

Calibration indicates that different combinations of quickflow, slowflow and streamflow attenuation coefficients can result in similar fits between observed and predicted stream nitrogen concentrations. This arises because ROTAN-Annual is 'over parameterised' in that three attenuation coefficients are calibrated using only stream nitrogen concentrations. If more groundwater information (groundwater concentrations, overlying land use and groundwater age) and/or streamflow attenuation (longitudinal profiles of flow and concentration) existed it might be possible to constrain the attenuation coefficients further.

Calibration also indicates that spatially uniform attenuation coefficients do not produce a satisfactory match between observed and predicted TN concentrations at all monitoring sites. There are two possible reasons.

1. The mismatch may result from errors in the input or monitoring data. For example, Calibration 1 implies high attenuation in the Awahou (AWA). The reason is that land use intensification during the 1970-1990s resulted in high nitrogen losses in the AWA during the calibration period, and to match observed stream concentrations the model invoked high attenuation. Were some of the nitrogen losses to have by-passed the monitoring site (e.g., to have flowed direct to the lake), or were nitrogen not to have reached the AWA monitoring site during the

calibration period (e.g., because groundwater travel times were longer than modelled), then calibration would have yielded lower attenuation coefficients.

2. There may be spatial differences in attenuation coefficient. For example groundwater denitrification (the likely mechanism for slowflow attenuation) is reported to vary according to the supply rate of dissolved organic carbon (Rivett et al. 2008). It is conceivable that carbon supplies and denitrification rates vary between aquifers, although very little information was located on differences in groundwater chemistry in the Rotorua basin. Morgenstern et al. (2004) reported results from synoptic sampling of 16 bores and springs in July 2003 in the Rotorua/Okareka catchments. In one bore (near the lake edge), oxygen and DIN concentration were very low and the authors suggest denitrification may have occurred. In the other bores and springs, oxygen concentrations were high (5-10 g m^{-3}), suggesting low denitrification rates in many parts of the catchment.

Differences between soils may also result in spatial variability in quickflow attenuation, but no soil attenuation data was located for the Rotorua region.

The use of two published estimates of groundwater mean residence time (Morgenstern et al. 2004 and 2015) caused differences in calibrated slowflow and streamflow attenuation, but steady-state lake loads that differed by only 4%.

ROTAN-Annual assumes that travel times from each REC sub-catchment vary with distance between the sub-catchment mid-point and where groundwater emerges into a stream or the lake. This assumption is untested and were travel times to be distributed differently within an aquifer (e.g., to vary with distance to some power as a result of sinuosity or variations in permeability), then attenuation coefficients and steady-state loads would differ from those presented here. Recently GNS developed a groundwater model of the Rotorua catchment (Daughney et al. 2015) that may allow refinement of flow pathways and travel times in ROTAN-Annual. Preliminary discussions have taken place with GNS but they were unable to provide additional information within the time frame of this study.

In the ROTAN-Annual model, predicted nitrogen concentrations in streams and the steady-state lake load were not sensitive to the choice of quickflow attenuation coefficient. This is because of the high drainage fraction in the majority of catchments (viz., very little nitrogen travels via the quickflow pathway). This finding is valid provided the drainage fractions used in ROTAN-Annual are accurate, and that water and nitrogen drain in the same proportions. The implication is that quickflow attenuation coefficients can be specified *a priori* and attention focused on estimating the slowflow and streamflow attenuation coefficients.

The slowflow and streamflow attenuation coefficients were found to be inversely correlated. Both coefficients strongly influence stream concentrations (whereas quickflow attenuation has only a minor impact). It is not surprising that across a number of calibrations, as one coefficient increased the other decreased. The steady-state lake load (assuming current land use) was found to be negatively correlated with both the slowflow and streamflow attenuation coefficients. As slowflow attenuation decreased, groundwater was predicted to make a progressively larger contribution to lake load. During the calibration period, streamflow attenuation increased to offset the decreasing slowflow attenuation so that observed and predicted stream concentrations matched from 1970-2015. However, beyond 2015 low slowflow attenuation resulted in high groundwater loads which were not fully offset by high streamflow attenuation, and consequently the steady-state lake load increased.

During calibration, constraints were placed on the three attenuation coefficients based on published attenuation coefficients and/or calculations using published data. Little published information could be located on slowflow attenuation in the Rotorua region (viz., nitrogen removal rates in groundwater) and so a wide range of values was used during calibration (range $0.0001 < \alpha < 0.01$ y^{-1}, Table 2-2, Table 2-3). This range furnished steady-state lake loads in the range $660 < SS < 860$ t y^{-1} and $390 < SS < 490$ t y^{-1} for current land use and nitrogen reductions respectively.

It is well known that 'hill climbing' and 'stepping' calibration methods may find local optima at the expense of missing other combinations of coefficients that fit observations. Starting searches with different initial coefficients reduces this risk, and the risk decreases as the number of starting combinations is increased (typically 1,000-10,000 starting combinations are used for complex models). The frequency distribution for steady-state lake load (Figure 3-8) is only approximate because it was derived using the results from only 1000 calibrations. However, increasing the number of calibrations would not necessarily improve the accuracy of predicted lake loads. The reason is that we do not know the 'true' distribution of the nitrogen losses and groundwater travel times or the 'true' errors in other model inputs (e.g., aquifer boundaries). Hence we cannot determine the 'true' distributions of the slowflow or streamflow attenuation coefficients from the available information.

There is no way of deciding solely by examining model calibration results which combinations of coefficients are truly optimal. Such a judgement needs to be based on other information (e.g., measured longitudinal profiles of stream flow and concentration and/or a more detailed examination of groundwater concentrations, overlying land use and groundwater age). Were either the slowflow or streamflow attenuation coefficient to be determined independently, then it would be possible to reduce the uncertainty in the other 'calibrated' coefficient(s) and hence reduce the uncertainty in predicted lake loads. Some comfort can be taken, however, from the fact that, when used to predict steady-state loads to the lake assuming constant land use, different combinations of coefficients derived during Calibration 2 gave results (700-808 t y^{-1}) that differed by only ±7% (Table 3-6).

If the main objective is to estimate how changes in farming practice and point source management affect the nitrogen load on the lake, then non-uniqueness of calibrated attenuation coefficients does not pose a severe problem (viz., the effects of changing nitrogen losses can be investigated by running the model using one or more combinations of attenuation coefficients with the reasonable expectation that predictions will be unbiased). However, non-uniqueness makes it impossible to decide, based solely on the modelling, what the best approach should be to intercepting nitrogen. Calibration provides no guidance on whether to target quickflow attenuation (e.g., by improving riparian buffer strips), slowflow attenuation (e.g., by enhancing groundwater denitrification) or streamflow attenuation (e.g., by enhancing aquatic plant growth). That decision needs to be made using other information.

Based on the synoptic sampling of Morgenstern et al. (2004) it was estimated that the slowflow attenuation coefficient may lie in the range $0.003 < \alpha < 0.0082$ y^{-1} (Table 2-3) (95% confidence interval). This range is smaller than was used during calibration ($0.0001 < \alpha < 0.01$ y^{-1}) and when used in scenario modelling reduced the uncertainty in predicted steady-state lake loads to $670 < SS < 840$ and $390 < SS < 460$ t y^{-1} (95% confidence interval) for current land use and loss reductions respectively.

Rutherford et al. (2011) estimated a steady-state lake load assuming constant land use of 725 t y^{-1} (excluding rainfall on the lake) which falls within the range from this study ($670 < SS < 840$ t y^{-1}). The earlier study did not investigate uncertainty in model calibration or the sensitivity of predictions to uncertainty because the ROTAN-2011 model does not lend itself to automatic calibration and sensitivity analysis. The current study found that the uncertainty in calibrated attenuation coefficients and steady-state loads was moderately high. Thus while the 'most likely' values of steady state load (750-760 t y^{-1}) may appear larger than the earlier estimate (725 t y^{-1}), the difference is probably not statistically significant.

The target load of 405 t y^{-1} (excluding rainfall on the lake) falls within the range estimated during this study ($390 < SS < 460$ t y^{-1}) but is lower than the 'most likely' values of 420-440 t y^{-1}. The difference (15-35 t y^{-1}) suggests that the loss reductions specified by BoPRC, while significantly reducing lake loads, may not quite reach the target of 405 t y^{-1}. There are two qualifications. Firstly, this study estimates the uncertainty in steady-state lake load to be 3-10% and for uncertainties near the upper bound the 'most likely' lake loads are not be significantly different from the target. Secondly, the target reductions for engineering and gorse control (total 80 t y^{-1}) (Table 3-9) were set based on ROTAN-2011 modelling which used OVERSEER v5. While engineering and gorse control measures targets were not based on OVERSEER modelling, the percentage reductions achieved through these measures is lower in this study (15%) than in the 2011 study (25%). Were these measures to total 25% of all reductions, engineering and gorse control reductions would each need to increase to 150 t y^{-1}.

ROTAN-Annual predicted a slower rate of recovery in lake load than an earlier (2011) study. The rapid decrease in lake load reported by Rutherford et al. (2011) was unexpected. It arose because aquifers with large MRTs (e.g., Hamurana, Waingaehe and Awahou) were modelled as fully-mixed reservoirs whose volumes were large. Their large volume 'buffered' the impacts of land use intensification and aquifer nitrogen concentrations increased only slowly. When loss reductions were modelled, the aquifers with small MRTs (e.g., Ngongotaha), which were modelled as fully-mixed reservoirs whose volumes were small, responded quickly and accounted for the rapid decrease in lake load. Questions were raised at the time about whether ROTAN-2011 realistically represented the behaviour of the larger aquifers, even though their MRTs matched published values.

The reason ROTAN-Annual predicted a slower recovery is that it modelled more aquifers with a wider range of MRTs than ROTAN-2011. For example, the Awahou catchment was modelled in ROTAN-2011 using 2 sub-aquifers in series both with MRTs of c. 30 years, whereas in ROTAN-Annual the catchment was modelled using 34 sub-aquifers in parallel with MRTs that varied from 9-112 years. The average MRT for the entire catchment was the same for both models. In ROTAN-Annual loss reductions in land close to the Awahou springs reduced stream concentrations quickly. However, loss reductions in land far distant from the springs took longer to affect stream concentrations than had been predicted by ROTAN-2011.

5 Acknowledgements

Alastair MacCormick (BoPRC) has made a major contribution to this study by collating and supplying information on farming systems, land use history and OVERSEER estimates of nitrogen losses. Neale Hudson (NIWA, Hamilton) and Andy Bruere (BoPRC) have guided the study, provided valuable input and asked appropriate 'difficult' questions.

6 References

Alexander, R.B., Elliott, A.H., Shankar, U., McBride, G.B. (2002) Estimating the sources and transport of nutrients in the Waikato River Basin, New Zealand. *Water Resources Research*, 38(12). 1268, doi:10.1029/2001WR000878, 2002.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M. (1998) Crop evapotranspiration: guidelines for computing crop water requirements. *FAO Irrigation and Drainage Paper* 56. Rome.

Daughney, C.J., Toews, M.W., Ancelet, T., White, P.A., Cornaton, F.J., Stokes, K., Maxwell, D., Tschritter, C. (2015) Implementing and evaluating models of steady-state groundwater flow under baseflow conditions, Lake Rotorua catchment, New Zealand. *GNS Science Report,* 2015/52. December 2015 : 128.

Environment Bay of Plenty (2007) Proposed Lakes Rotorua & Rotoiti Action Plan written by Environment Bay of Plenty, Rotorua District Council and Te Arawa Lakes Trust. *Environmental Publication,* 2007/11.

Environment Bay of Plenty (2009) Lakes Rotorua & Rotoiti Action Plan written by Environment Bay of Plenty, Rotorua District Council and Te Arawa Lakes Trust. *Environmental Publication,* 2009/03.

Fish, G.R. (1975) Lakes Rotorua and Rotoiti, North Island, New Zealand: their trophic status and studies for a nutrient budget. *Fisheries Research Bulletin,* 8. Ministry of Agriculture & Fisheries, Wellington.

Hoare, R.A. (1980b) The sensitivity to phosphorus and nitrogen of Lake Rotorua, New Zealand. *Progress in Water Technology,* 12: 897–904.

MacCormick, A. (2016) Methodology used to transition ROTAN Overseer v5 files into OVERSEER v 6.2.0 and integrate it into benchmarking data. *Draft BoPRC report 16th May 2016.*

Morgenstern, U., Gordon, D. (2006) Prediction of Future Nitrogen Loading to Lake Rotorua. *GNS Science Report* 2006/10. Geological & Nuclear Sciences, Lower Hutt.

Morgenstern, U., Reeves, R., Daughney, C., Cameron, S. (2004) Groundwater age, time trends in water chemistry, and future nutrient load in Lake Rotorua and Okareka Area. *Institute of Geological & Nuclear Sciences, Client Report* 2004/17.

Morgenstern, U., Daughney, C.J., Leonard, G., Gordon, D., Donath, F.M., Reeves, R. (2015) Using groundwater age and hydrochemistry to understand sources and dynamics of nutrient contamination through the catchment into Lake Rotorua, New Zealand. *Hydrology & Earth Systems Science, 19:* 803-822.

Rivett, M.R., Buss, S.R., Morgan, P., Smith, J.W.N., Bemment, C.D. (2008) Nitrate attenuation in groundwater: A review of biogeochemical controlling processes. *Water Res.* 42: 4215-4232.

Rutherford, J.C., Tait, A., Palliser, C.C., Wadhwa, S., Rucinski, D. (2008) Water balance modelling in the Lake Rotorua catchment. *NIWA Client Report* HAM2008-048. Hamilton.

Rutherford, J.C., Palliser, C.C., Wadhwa, S. (2009) Nitrogen exports from the Lake Rotorua catchment – calibration of the ROTAN model. *NIWA Client Report* HAM2009-019. Hamilton.

Rutherford, J.C., Palliser, C.C., Wadhwa, S. (2011) Prediction of nitrogen loads to Lake Rotorua using the ROTAN model. *NIWA Client Report* HAM2010-134. Hamilton.

Rutherford, J.C. (2013) Effects of land use on nutrients. Phase 2 studies in the Tukituki River, Hawke's Bay. *NIWA Client Report* HAM2013-025. Hamilton.

Rutherford, J.C., Palliser, C.C. (2014) Lake Rotorua catchment boundaries. Phase 2 water budget. *NIWA Client Report* HAM2014-014. Hamilton.

Selbie, D.R., Watkins, N.L., Wheeler, D.M., Sheperd, M.A. (2003) Understanding the distribution and fate of nitrogen and phosphorus in OVERSEER. In: *Proceedings of the New Zealand Grassland Association,* 75: 103-112.

Viner, A.B. (Ed) (1987) Nitrogen, phosphorus and oxygen dynamics in rivers. In: Inland Waters of New Zealand. *DSIR Bulletin* 241.

White, P.A., Cameron, S.G., Kilgour, G., Mroczek, E., Bignall, G., Daughney, C., Reeves, R.R. (2004) Review of groundwater in the Lake Rotorua catchment. *GNS Client report,* 2004/130: 231.

White, P.A., Rutherford, J.C. (2009) Groundwater catchment boundaries of Lake Rotorua. *GNS Science report, 2009/75LR for Environment Bay of Plenty.*

White, P.A., Tschritter, C., Lovett, A., Cusi, M. (2014) Lake Rotorua catchment boundary relevant to Bay of Plenty Regional Council's water and land management policies. *GNS Science Consultancy Report,* 2014/111: 99.

Williamson, R.B., Cooke, J.G. (1982) Water quality of the Waiohewa Stream, Rotorua. *New Zealand Journal of Marine & Freshwater Research,* 16: 327–337.

Williamson, R.B., Smith, C.M., Cooper, A.B. (1996) Watershed riparian management and its benefits to a eutrophic lake. *Journal of Water Resources Planning and Management,* 122: 24–32.

Zhang, L., Dawes, W.R., Walker, G.R. (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research,* 37(3): 701–708.