

Estimation of nutrient sources and transport for New Zealand using the hybrid mechanistic-statistical model SPARROW

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Abstract

The hybrid mechanistic-statistical catchment model SPARROW was applied to predict the mean annual load of nitrogen and phosphorus in streams throughout New Zealand (270,000 km²). The loads from land areas, point sources, and erosion are routed through the drainage network (576,300 reaches) with first-order stream decay and attenuation in lakes and reservoirs. Model parameters were determined by calibration against loads measured in the national water quality network (77 sites). For nitrogen, the model predicted the measured loads well (R^2 of 0.956 and RMSE of 0.33 in natural-log space), while for phosphorus the model fit was not as good (R^2 of 0.900 and RMSE of 0.58). The predictions of exported yields for streams with catchments >20 km² are broadly comparable with previous compilations of yields for various land-use classes for nitrogen, but are larger than the previous measurements for phosphorus. The calibrated stream attenuation and lake/reservoir rates were broadly consistent with previous measurements. The predicted load of total nitrogen (TN) delivered to the coast was 167,700 t yr⁻¹, which is 45% of the loads entering the streams. For total phosphorus (TP) the predicted load to the coast was 63,100 t yr⁻¹, 44% of the load

entering the streams. Reservoir/lake attenuation makes a relatively small contribution to the overall attenuation compared with in-stream attenuation (3.5% for nitrogen and 8.5% for phosphorus). The largest contribution of total nitrogen is from pastoral land uses, together accounting for 70% of the total nitrogen load to the coast. Land used for dairying makes a disproportionately large contribution to the load of total nitrogen in relation to the area of land (37% of the load versus 6.8% of the land). For total phosphorus, the highest contribution of the load to the coast is from erosion (53.2%). Point sources contribute only a small proportion of the load to the coast (3.2% for nitrogen, 1.8% for total phosphorus). The monitoring network does not include streams with catchments smaller than 10 km², so model predictions for streams smaller than 10 km² should be used with caution.

Keywords

Catchment model, nitrogen phosphorus, SPARROW, New Zealand

Introduction

The quantity of nutrients in New Zealand streams is of interest for predicting the

growth of nuisance plants in streams and eutrophication of lakes and coastal areas (e.g., James *et al.*, 2001). A particular concern in New Zealand is the influence of diffuse sources of nutrients associated with pastoral agriculture, and potential changes in the load of nutrients as pastoral land use becomes more intensive (Wilcock, 1986), especially as the relative contribution from point sources is diminishing (Hickey and Rutherford, 1986).

Typical nutrient yields for various land uses in New Zealand have been assessed in the past from studies of small to medium-sized catchments (0.1–100 km²) (Elliott and Sorrell, 2002; Wilcock, 1986). These yields have been used in conjunction with estimates of point-source loads to assess the load of nutrients to waterways for the country as a whole (Rutherford *et al.*, 1987; Taylor *et al.*, 1997), and are used by water managers for

estimating the loads of nutrients to lakes and waterways. The variability in yield for any given land use, however, is unresolved, leading to difficulties in selecting a suitable yield to use for any given location or catchment and uncertainties in the resulting estimates of load.

The New Zealand National Rivers Water Quality Network (NRWQN, Fig. 1) is a network of 77 water quality monitoring stations established in 1989 to detect trends in water quality, develop a better understanding of the nature of the water resources, and assist with water management (Smith and Maasdam, 1994). Statistical analysis of the concentration data from this network and other one-off national surveys of concentrations show water quality is influenced by the degree of pasture development in the catchment, flow rate, catchment lithology, elevation and

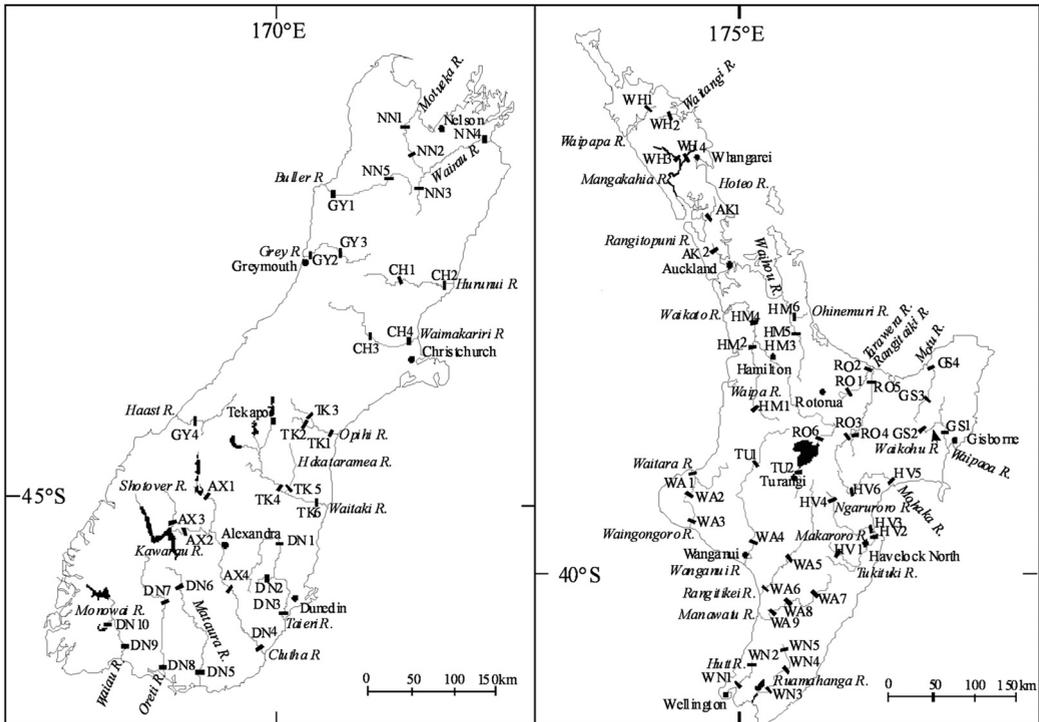


Figure 1 – National Rivers Water Quality Network (NRWQN) monitoring site locations and major rivers.

catchment slope (Close and Davies-Colley, 1990; Maasdam and Smith, 1994). However, the National Rivers Water Quality Network data have not yet been used to assess the load of nutrients, or the relation of nutrient load to land use, which is of particular interest for the management of water quality in lakes and estuaries.

In this paper the loads from the National Rivers Water Quality Network sites are used to calibrate a national-scale model of in-stream loads of total nitrogen (TN) and total phosphorus (TP). Particular questions of interest are: how nutrient loads relate to diffuse and point sources; predictions of loads for ungauged streams; predictions of the effect of changes in land use; and prediction of the national load of nutrients to the coastline.

In this study we used an advanced regional regression technique, SPARROW (SPATIally-Referenced Regression On Watershed attributes), to model the loads (Alexander *et al.*, 2002; Smith *et al.*, 1997). The model has been used to assess nutrient loads for the United States of America and regions therein (McMahon *et al.*, 2003; Moore *et al.*, 2004; Preston and Brakebill, 1999; Smith *et al.*, 1997). Applying this model to the 13,900 km² Waikato river basin in New Zealand (Alexander *et al.*, 2002) gave predictions of nutrient yield that were typically within 30% of the measured yield, showed a strong influence of land use and attenuation in reservoirs and streams, and gave parameter values consistent with published values. This successful trial application prepared the way for a full national application (270,000 km²), including catchments with a wider range of climate, geology, and topography and a finer spatial scale for the reach network and associated subcatchments.

SPARROW is a form of nutrient load model that is similar to conventional regional regression—it is statistically based and uses data from a number of monitoring stations to calibrate the model parameters. But it

goes beyond that to include components of distributed mechanistically-based catchment modelling such as mass balance, stream transport processes, and the spatial location of diffuse and point sources within the catchment and drainage network. The mathematical description of these components is formulated in a physically plausible manner, so that their coefficients are physically interpretable. The conventional regional regression attributes allow statistical interpretation of model parameters and analysis of the information in data from a number of sites simultaneously. The mechanistically-based and spatially-referenced components allow interpretation of model parameters. This provides a conceptually sound basis for accounting for attenuation, making good use of spatial information such as variations in topography and location of sources in relation to the stream network. The model processes and source terms are kept simple and only the annual average nutrient load is calculated, so large-scale models are easier to set up and quicker to run, compared with more mechanistically-based continuous-time models (e.g., AGNPS, Young *et al.*, 1995).

Methods

Model description

Earlier versions of the SPARROW model have been described in detail previously (Alexander *et al.*, 2002, 2004; Smith *et al.*, 1997). Here we briefly highlight the key components of the model structure and describe some key changes from earlier versions. The conceptualisation of the model is shown in Figure 2, while Table 1 gives the key terms.

Within each subcatchment or incremental watershed, the load of nutrient generated for source type *n* is the product of the amount of source (area of land use, emitted load from the point sources) times a source coefficient (yield for diffuse sources; a dimensionless

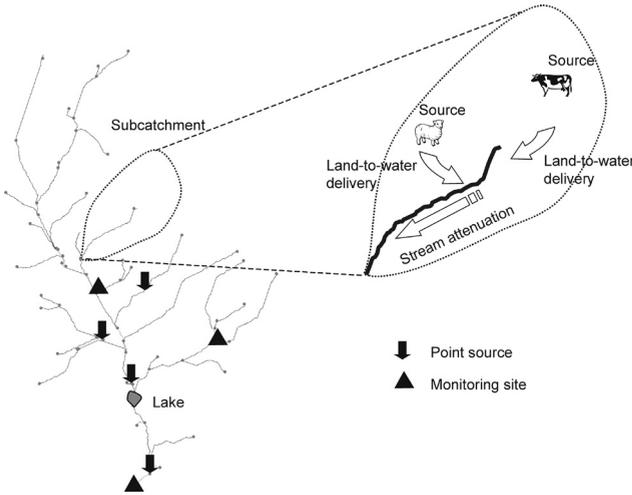


Figure 2 – Schematic representation of SPARROW sources and transport.

coefficient for point sources). This source load is then modified by a land-to-water delivery term, which is an exponential function of a number of delivery variables (such as rainfall or land drainage class). These modified sources are then summed for a given

subcatchment to give the total load entering the associated stream reach. In-stream decay is modelled by a first-order decay term, and the load is then accumulated and attenuated during movement down the stream network. Nutrients entering the reach from the incremental watershed are assumed to be subject to decay over half of the reach length on average, while loads entering from upstream reaches are subject to the decay over the full reach. A pre-determined calculation sequence ensures that before the load in a particular reach is calculated, the load for all upstream reaches is

already calculated. The load is attenuated using a separate decay function for lakes or reservoirs.

If a monitoring station is encountered at downstream points, then the loads are set to the monitoring station load. The model

Table 1 – Main terms and symbols in the model

Term	Equation	Symbols
Source terms (t/yr)	$\beta_n S_n$	S_n = quantity of source type n. For diffuse sources, area of land use (km^2). For point sources, load emitted by the source (t yr^{-1}) β_n = source coefficient for source type n. For diffuse sources, a yield ($\text{t km}^{-2} \text{yr}^{-1}$). For point sources, a dimensionless source coefficient
Land-to-water delivery factor	$\exp(-\alpha' Z_j)$	α = vector of delivery coefficients Z_j = vector of delivery variables for subcatchment j
Attenuation factor for a stream reach	$\exp(-\delta_m^s L_m)$	δ_m^s = first-order decay coefficient (km^{-1}) for stream reach m L = length of stream reach m
Attenuation factor for a reservoir	$(1/(1+\lambda^r q_l^{-1}))$	λ^r = reservoir settling velocity (m yr^{-1}) q_l = overflow rate (outflow /area) (m yr^{-1}) for reservoir l

calibration is thus sensitive to the incremental load and decay between monitoring stations, which improves the ability of the model to estimate the decay parameters.

We allowed the stream decay coefficient δ^s to vary with the flow in two ways. In the step-wise model, several discrete flow classes were defined and a different value of δ^s was used for each flow class. The cut-off flow between classes was determined by trial and error, while the values of δ^s were treated as calibration parameters. In the second approach, the continuous model, a continuous function of flow was used, as suggested by Cooper and Bottcher (1993):

$$\delta^s = aQ^B \quad (1)$$

where a is the continuous decay coefficient and B is the continuous decay exponent, which are fitted parameters.

The model calculations are performed using the statistical software package SAS (SAS Institute Inc., Cary, North Carolina, USA), with extensive use of the Interactive Matrix Language (IML) module. The model is calibrated using the non-linear Levenberg-Marquardt least-squares optimisation procedure within IML to minimise the sum of squares of differences between the logarithms of the observed and predicted mean annual loads at the monitoring stations (Root Mean Square Error – RMSE). The loads are log-transformed to ensure that high loads do not unduly influence the optimisation, and this also makes the residual errors closer to normally-distributed, which improves the robustness of statistical estimates such as confidence intervals for the parameters. The bootstrap estimation capability of SPARROW (Smith *et al.*, 1997) was not used due to the large disk storage requirement and large computational times for non-parametric bootstrapping (where repeat optimisations are performed). We therefore assume approximate normality in the distributions of the parameters and residuals.

The model architecture and computer software have been recently revised (Alexander *et al.*, 2004) to accommodate, among other things, a more flexible method for load accumulation and decay in the network.

Data sources

The input data (excluding model parameters) are summarised in Table 2 and are described in more detail below.

Monitoring station measured loads

The mean annual loads of total nitrogen (TN) and total phosphorus (TP) were determined for the 77 sites in the National Rivers Water Quality Network (Smith and Maasdam, 1994), using the same rating-curve methods as in the earlier Waikato application (Alexander *et al.*, 2002). Samples are taken from these sites monthly. We chose a four-year period rather than a longer period to avoid complications of changing land use in some of the catchments. The period 1996-1999 was chosen, as the data on land use was based on satellite imagery from 1996. For each site, a regression was fitted to the logarithm of the instantaneous flux data (including a seasonal term, a linear temporal trend term, and the logarithm of the flow) and this regression was applied to hourly flow records to give the load over the four-year period. The flows used in these calculations were obtained from continuous flow records available from rated flow gauges for all sites. A smearing-method correction factor was applied to account for log-transformation bias error (Duan, 1983).

The mean proportion of developed pasture in the catchments of the monitored locations is 34.7%, which compares with the 38.6% of New Zealand that is in pasture. There is a wide range of pasture in the catchments (lower and upper deciles of 1.8% and 74.8% respectively). Hence the monitoring stations represent a cross-section of this land use, which is expected to make a large contribution to nutrient loads. The catchment areas of

Table 2 – Summary of input data and calibration parameters. References to data sources are given in the main text.

Input data	Data Source
Monitoring station	Rating-curve methods applied to 77 sites in the NRWQN from 1996-1999
Drainage network, reach lengths subcatchments boundaries, and subcatchments areas	Automated catchment delineation using DEM based on 20 m contours, with modification for flow diversions
Lakes location and area	Digitised 1:50,000 topographic maps, lakes < 10,000 m ² removed
Land use	Pasture, tussock, native forest, scrub, exotic pine plantations, water bodies, urban land use, and other land use, derived from Landcover Database digital maps. Subdivision of pasture into dairy, beef, and other pasture derived from Agribase digital maps of pastoral land use.
Point source location and load	Local council data, measured or estimated loads
Sediment	Sediment yield map (Hicks and Shankar, 2003)
Mean annual rainfall	Digitised maps of rainfall normals
Soil erosion potential	Soil erosion severity class from the Land Resources Inventory
Soil drainage	Soil internal drainage class from the Land Resources Inventory
Catchment slope	Mean value of local slope, obtained from 30 m DEM derived from 20 m contours
Mean annual air temperature	Land Environments of New Zealand climate surface
Stream mean annual discharge	Runoff depth times catchment area, accumulated down network, where runoff is estimated from the mean annual rainfall minus estimated actual evapotranspiration

the monitoring stations vary considerably, ranging from a minimum of 13.7 km² to a maximum of 20,500 km², with a median of 1130 km².

The calculated nutrient yields (load divided by catchment area) range over 2 orders of magnitude (Fig. 3), and there is also a wide range of water yield. The median standard error of the natural logarithm of the calculated load is 0.32 for total nitrogen (38% error), while the value for total phosphorus is higher (0.47, or 60% error). The mean

yield is 5.33 kg ha⁻¹ yr⁻¹ for total nitrogen and 1.23 kg ha yr⁻¹ for total phosphorus.

Drainage network and subcatchments

The drainage network was derived from a 30 m digital elevation model (DEM), applying standard D-8 flow accumulation and catchment delineation techniques within Arc/Info (ESRI, 1992). The methods are described in more detail in Shankar *et al.* (2002). The DEM was based on 20 m elevation contours, and stream locations

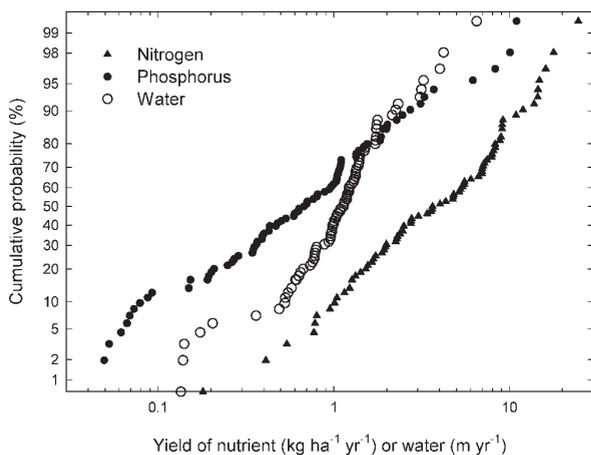


Figure 3 – Cumulative probability of nitrogen, phosphorus and water yields at the monitoring stations in the National Rivers Water Quality Network (NRWQN). The probabilities are plotted on a normal probability scale.

as identified in digitised 1:50,000 scale topographic maps were ‘burned in’ to the DEM. The drainage network was modified manually to account for diversions of water for hydroelectric power schemes.

The resulting drainage network has 576,300 reaches, each with an associated subcatchment. The mean subcatchment area is 0.46 km² and the median area is 0.36 km². The mean reach length is 740 m, with a median of 530 m. 51% of the subcatchments are headwater catchments.

The location and area of reservoirs and lakes were determined from digitised 1:50,000 scale maps. Lakes smaller than 10,000 m² were removed, so that eventually the model had just 2,585 lakes, compared with 54,500 lakes nationally. The smaller lakes amount to only 3% of the total lake area, so neglecting them will have an insignificant effect on the overall lake attenuation. Reaches with their centre-line within the 2,585 lakes were identified as special reaches in which no stream attenuation takes place. Nutrient attenuation is applied at the outlet reach of the lake/reservoir.

Source terms

Several types of land use were identified for use as diffuse-source terms in the model. The area of each land use within each subcatchment was derived from the Landcover Database (obtained from Terralink International Limited), which is based on image analysis of SPOT satellite images (primarily in 1996) and has a 1 ha minimum mapping unit area. The land-use classifications in the Landcover Database were reclassified to give the following categories: pasture, tussock, native forest, scrub, exotic pine plantations, water bodies, urban land use, and other land use. Various combinations of land uses were also investigated (for example, lumping native forest, pine, and scrub into a category of trees).

Pasture land use was further broken down into beef, dairy, and non-beef/dairy pasture (e.g., sheep), using the 1999 Agribase national database of farm land use, which is maintained by AgriQuality New Zealand Ltd. These maps give the type of pasture land use for each cadastral lot, based on farmer surveys. More detailed information such as stocking rates, which is contained in the database, was not used due to difficulties in accessing the data.

Point-source locations and loads were obtained from local councils. Dairy farm milking shed discharges and discharges directly to coastal areas were excluded. The measured load was used if available (from samples of flow rate and concentration, usually taken as part of consent monitoring). If load measurements were not available, then the load was estimated from the measured flow rate and estimated concentrations (from other sites with a similar waste type, or using typical concentrations for the waste type from sources such as Hickey *et al.*, 1989) or the population and the waste type. The data

sources are too numerous to list individually. The maximum permitted (consented) discharge rate was not used, as this often differs from the actual discharge. Point sources applied to land were included. The largest 126 point sources were selected, leaving out smaller sources such as rural holiday camps. The larger sources included municipal sewage discharges, piggeries, abattoirs, dairy factories, and wood processing plants.

For the phosphorus model, an erosion source term was investigated. Some areas of New Zealand have very high erosion rates due to high rainfall, soft rocks, and steep slopes. The data on phosphorus yields suggest that yields are higher in areas of high sediment load. It would be difficult to account for these in a single baseline phosphorus yield. Therefore we used a separate assessment of sediment yields (Hicks and Shankar, 2003) that incorporates the effects of rainfall and geology and was available in electronic format.

Land-to-water delivery variables

Several land-to-water delivery factors were investigated for the calibration. For each subcatchment, an area-weighted mean value of each variable was determined. Mean annual precipitation was obtained from digitised maps of rainfall normals (Tomlinson, 1994). The soil erosion severity index (6 classes, where class 0 is for the lowest severity) and soil internal drainage class (5 classes, where class 1 is for the worst-drained soils) were obtained from the New Zealand Land Resources Inventory (Newsome *et al.*, 2000). The slope was determined from the gradient of the 30-m DEM used for derivation of the drainage network. The mean annual air temperature was determined from the Land Environments of New Zealand climate surface (Leathwick, 2002).

All the delivery variables were mean-adjusted by subtracting the mean value over

the whole of the country. Hence the land-to-water delivery term is equal to unity if the variable is equal to the mean value. This is to help in interpreting the diffuse source coefficients—the values of the source coefficients are then standardised so that they relate to the mean values of the delivery variables.

Mean stream flow

Mean annual stream flow is used in estimates of in-stream and reservoir decay. The flow generated by each subcatchment was determined from the subcatchment area times the water yield. The yield was determined from the mean annual precipitation (as described above) minus the estimated evapotranspiration. The evapotranspiration was determined from a surface of Penman pasture evapotranspiration potential (Leathwick, 2002) times a correction factor based on evapotranspiration estimates from a daily water balance model (New Zealand Meteorological Service, 1986; Zhang *et al.*, 2004). These incremental flows were accumulated down the stream network to give the total flow in each reach.

Modelling and calibration strategy

A range of model variants (Table 3) was investigated—each variant included a different combination of model terms. The parameters associated with the terms were determined through the SPARROW automated calibration capabilities, with values that were unconstrained except for the reach attenuation coefficients (discussed later). We used a step-wise addition strategy for most model terms, in which we started with a simple model with few terms (combining land uses where appropriate). Additional terms were added and retained if they improved the model (reduced the RMSE of the model and the p values of the parameters and, in the case of land-use classes, the source coefficient

was distinct from other land uses). For flow classes in the attenuation model, we started with a large number of flow classes and then reduced the number by combining adjacent flow classes with similar coefficients or unstable coefficients (oscillating function of flow), provided that this did not increase the RMSE substantially. Influential outliers were removed after we investigated the associated data. As a final stage, we tried simplifying the

model through a step-wise removal of terms. The terms retained in the final models are shown in Table 3 and are discussed in more detail below.

Results

Due to the large number of subcatchments, model runs took typically 30 minutes on a 1.8 GHz Pentium PC for a model with 10 parameters. The optimisations converged

Table 3 – Summary of model terms and parameters investigated and included in the final model forms.

Model term	Parameter and units	Retained in final TN model	Retained in final TP model
Sources			
Point source	Point source coefficient (dimensionless)	Yes	Yes
Dairy pasture land use	Yield coefficient (kg ha ⁻¹ yr ⁻¹)	Yes	Yes
Beef pasture	As above	Yes ^a	Yes ^a
Remaining pasture classes	As above	Yes ^a	Yes ^a
Tussock	As above	Yes ^c	Yes ^d
Native forest	As above	Yes ^b	Yes ^d
Scrub	As above	Yes ^b	Yes ^d
Exotic pine plantations	As above	Yes ^b	Yes ^d
Water bodies	As above	Yes ^c	Yes ^d
Urban land use	As above	Yes ^c	Yes ^d
Other land use	As above	Yes ^c	Yes ^d
Sediment	Sediment coefficient (g kg ⁻¹)		Yes
Land-to-water delivery			
Rain (m ⁻¹)	Rain coefficient (m ⁻¹)	Yes	Yes
Erosion potential	Erosion potential coefficient (per index)		
Soil drainage	Drainage coefficient (per drainage index)	Yes	
Catchment slope	Slope coefficient (per degree)		
Air temperature	Temperature coefficient (per degree Celsius)		
Aquatic loss			
Stream decay, step-wise function	Decay coefficients (km ⁻¹), number of flow classes, and flow cutoff values	Yes	Yes
Stream decay, continuous function of flow	Constant and exponent for equation 1		
Reservoir decay	Reservoir settling velocity (m yr ⁻¹)	Yes	Yes

^aCombined into other pastures
^cCombined into other non-pasture

^bCombined into trees
^dCombined into non-pasture

rapidly (typically within 20 iterations), and the model was insensitive to variations of the initial estimates of the parameters. We will present the results for total nitrogen first, followed by total phosphorus.

The terms and parameter values for the final total nitrogen model (with stream attenuation a step-wise function of flow) are shown in Table 4. One outlier, the Hakataramea site (site TK5), was removed from the set of monitoring stations, as it had a high residual and was the only site that didn't fit the normal distribution of residuals. The data for this site were checked, but no reason has been found for the unexpectedly low load at this site.

The source terms retained in final total nitrogen model were point sources, areas of dairy pasture, other pasture (pasture area minus dairy area), trees (combined native and exotic forest plus scrub), and 'other

non-pasture' (tussock, water bodies, urban, cropping, bare ground) land uses. The 'other non-pasture' term was retained, even though its p value was above the traditional 0.05 level, because the coefficient is significantly different from that for trees and the overall model is improved by including this term.

The point-source term was retained in the total nitrogen model, despite its large uncertainty, to make the specification of the sources 'complete' and because point sources are of particular interest, especially close to the source. The drainage class and mean annual rainfall were retained as land-to-water delivery variables in the final total nitrogen model.

Plots of predicted versus observed load and yield are shown in Figure 4 for total nitrogen. The plots for total phosphorus are similar, but have greater scatter. The residual errors were approximately normally

Table 4 – Nitrogen model terms and coefficients

Coefficient	Value	Standard Error	p value ^d
Sources, β			
Point source coefficient (dimensionless)	1.38	0.76	0.073
Dairy pasture land use yield (kg ha ⁻¹ yr ⁻¹)	71.4	14.3	<0.001
Trees land use yield ^a (kg ha ⁻¹ yr ⁻¹)	5.87	1.05	<0.001
Other pasture land use yield (kg ha ⁻¹ yr ⁻¹)	18.2	4.07	<0.001
Other non-pasture land use yield (kg ha ⁻¹ yr ⁻¹)	0.830	0.454	0.072
Land-to-water delivery, α			
Drainage term (per drainage index) ^b	-0.238	0.092	0.012
Rain term (m ⁻¹) ^c	0.243	0.097	0.015
Aquatic loss			
Decay coefficient for flow class 1 ($Q < 0.1 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹)	0.335	0.180	0.067
Decay coefficient for flow class 2 ($0.1 < Q < 1 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹)	0.0917	0.0377	0.018
Decay coefficient for flow class 3 ($1 < Q < 10 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹)	0.0245	0.0090	0.008
Decay coefficient for flow class 4 ($Q > 10 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹) ^e	0	–	–
Reservoir settling velocity (m yr ⁻¹)	12.6	3.9	0.002
Root mean square error (natural log space)	0.33		
Adjusted R-squared	0.956		

^a Trees land use is the sum of exotic, indigenous, and scrub land uses. It is constrained to be non-negative

^b Mean adjusted drainage index used in the regression. The mean drainage index is 4.18.

^c Mean adjusted drainage index used in the regression. Mean rainfall is 1.855 m.yr⁻¹

^d The p value is based on a two-sided t-test of the parameter being equal to zero

^e Constrained to be non-negative

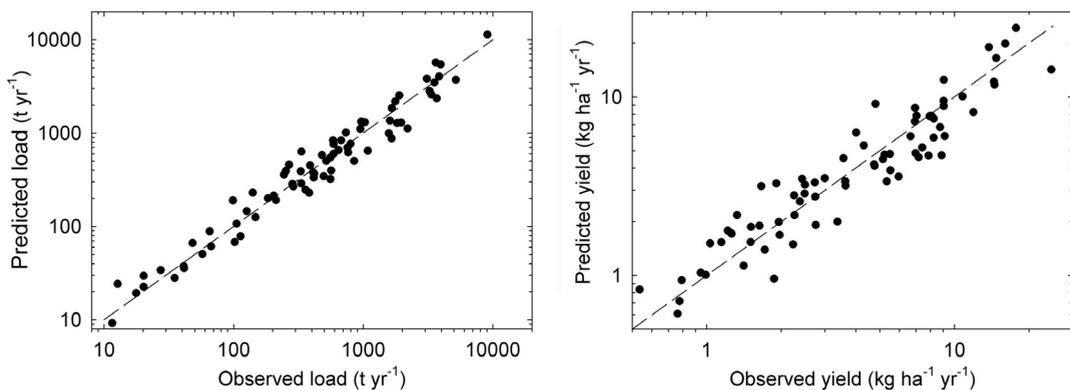


Figure 4 – Total nitrogen (TN) predictions versus observations: (a) load, (b) yield. The dashed lines are lines of perfect agreement.

distributed (in logarithmic space), there was no spatial bias in the errors, and the errors were approximately homoscedastic. The RMSE for the total nitrogen model was 0.33 (equivalent to 39% error), 74% of the predicted loads lie within 50% of the measured values, and 97% of the predicted loads (all but 2 sites: 3 if the Hakataramea outlier is included) lie within a factor of 2 of the measured values.

Four flow classes were used in the total nitrogen model. Addition of a further class for smaller flows ($<0.1 \text{ m}^3 \text{ s}^{-1}$) gave a coefficient similar to that for the next size class up ($0.1 < \text{flow} < 1 \text{ m}^3 \text{ s}^{-1}$) and also increased the error associated with that flow class without decreasing the RMSE, so the smaller flow class was not included in the final model form. Including additional intermediate flow classes tended to make the decay coefficients vary non-monotonically with flow, and degraded the significance of the decay coefficients. Reducing the number of flow classes further (to less than four) resulted in deterioration of the overall model (increasing RMSE). The coefficient for the largest flow was constrained to be positive, and the calibrated value was against this constraint. Otherwise, a small negative value was obtained for the decay coefficient (-0.001 km^{-1})—it is not statistically sig-

nificant, is of no practical significance, and it degrades the significance of the other terms in the model. Most streams in New Zealand have less than 150 km of length with $>10 \text{ m}^3 \text{ s}^{-1}$ flow.

The soil drainage class delivery term improved R^2 from 0.946 (no delivery terms) to 0.953 (drainage class only) and the RMSE from 0.36 to 0.34, and was retained in the model. Inclusion of slope, soil erosion severity index, and mean annual air temperature did not improve the model. Adding a term for rain delivery, in addition to the drainage term, improved R^2 to 0.956 and the associated coefficient was statistically different from zero, so the rain term was retained. A model with only a rain delivery term gave an R^2 of 0.953.

A total nitrogen model in which the in-stream decay coefficient is a continuous function of flow had slightly lower RMSE than the step-wise decay model. We chose to use the step-wise model, as the model residuals plot for the continuous model showed some systematic variation with measured load, and the final total phosphorus model was also step-wise.

The model terms and coefficients for total phosphorus are shown in Table 5. As with the total nitrogen model, the Hakataramea

Table 5 – Phosphorus model terms and coefficients

Coefficient	Value	Standard Error	p value ^b
Sources, β			
Point source coefficient (dimensionless)	1.85	1.48	0.215
Dairy pasture land use yield (kg ha ⁻¹ yr ⁻¹)	7.81	3.32	0.022
Other pasture land use yield (kg ha ⁻¹ yr ⁻¹)	4.36	1.89	0.023
Non-pasture land use yield (kg ha ⁻¹ yr ⁻¹)	0.508	0.307	0.102
Sediment (g kg ⁻¹)	0.255	0.102	0.015
Land-to-water delivery, α			
Rain coefficient (m ⁻¹) ^c	0.631	0.284	0.021
Aquatic loss			
Decay coefficient for flow class 1 ($Q < 1 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹)	0.196	0.102	0.060
Decay coefficient for flow class 2 ($1 < Q < 10 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹)	0.049	0.019	0.012
Decay coefficient for flow class 3 ($Q > 10 \text{ m}^3 \text{ s}^{-1}$) (km ⁻¹) ^a	0	–	–
Reservoir settling velocity (m yr ⁻¹)	132	45	0.005
Root mean square error (natural log space)	0.58		
Adjusted R-squared	0.900		

^a Constrained to be non-negative

^b The p value is based on a two-sided t-test of the parameter being equal to zero

^c Mean rainfall is 1.855 m.yr⁻¹

site was removed as an outlier. An additional site, the Waikohu (site GS2), was also removed as an influential outlier. The large sediment input at that site has a large error (the map-derived value is inconsistent with the measured sediment load). A step-wise decay coefficient model was used, with three flow ranges. Adding further flow ranges did not improve the model, and resulted in large p values for the coefficients. The non-pasture land uses were combined into a single class, as a breakdown into further classes did not improve the model or result in distinct source coefficients. The pasture land use was broken into dairy and other pasture, as this improved the R², even though the coefficients were marginally distinct. The erosion source term was retained in the model, as it had a highly significant coefficient and improved the R² from 0.84 to 0.90. The rainfall was the only land-to-water delivery term retained in the model, and this term was not applied to the erosion source, as that source should already incorporate delivery effects. As with total nitrogen, the model residual errors were approximately normally distributed,

there was no spatial bias, and the errors were approximately homoscedastic.

Discussion of the nitrogen model

Overall model performance

The R² value and RMSE of the total nitrogen model compare favourably with the values obtained from previous applications of SPARROW (Table 6). The RMSE is comparable to the estimated error in the measurements of the nitrogen load, suggesting that further reduction of the model RMSE would be difficult unless more refined load measurements are available—further model refinement would otherwise largely be a matter of resolving the measurement errors.

Sources and land-to-water delivery

The yields of contaminants entering the streams for various land uses after the land-to-water delivery factors are applied are shown in Table 7. These are comparable to the yield coefficients (β), but have a spread of values due to the variation in land-to-water variables

Table 6 – Comparison between performance of the New Zealand and other applications of SPARROW, in terms for adjusted R² value and root mean square error of residuals in log space (RMSE).

Study	TN		TP	
	R ²	RMSE	R ²	RMSE
New Zealand national (this study)	0.956	0.33	0.900	0.58
Waikato (Alexander <i>et al.</i> , 2002)	0.968	0.38	0.968	0.39
USA National (Alexander <i>et al.</i> , 2004; Smith <i>et al.</i> 1997; Alexander <i>et al.</i> , 2000)	0.882	0.43	0.82	0.71
Chesapeake Bay (Preston and Brakebill, 1999)	0.961	0.41	–	–
New England (Moore <i>et al.</i> , 2004)	0.95	0.41	0.94	0.48
Mississippi River Basin (Alexander <i>et al.</i> , 2000)	0.881	0.43	–	–

(mean annual rain and soil drainage). Whereas these SPARROW yields relate to the load entering streams, previous summaries of measured yields for New Zealand (Elliott and Sorrell, 2002) relate typically to streams with catchments larger than 10 km². We would expect stream attenuation to reduce the yields between these two scales. To gauge the importance of this effect, the SPARROW predictions were analysed to give the load leaving a reach as a fraction of the load entering the streams in the catchment of each

reach. The ratios were put into small bins of catchment area, and the median value for each bin was plotted against catchment area (Fig. 5). This demonstrates that as the scale of observation increases (as the catchment become larger), the exported yields decrease. We also used the SPARROW outputs directly to obtain the yield for catchments between 20 and 30 km² for catchments dominated by the various land-use classes (Table 7). The yields obtained in this manner are comparable to the values reported in

Table 7 – Comparison between yields (kg ha⁻¹ yr⁻¹) for SPARROW and compilations of yields previously measured in New Zealand. Land uses are defined by having at least 85% of the land in the specified land use. The other New Zealand studies are from Elliott and Sorrell (2002).

Land use	SPARROW median to streams (inter-quartile range)	SPARROW median exported yield, catchments 20-30 km ² (inter-quartile range)	Other NZ studies median (standard deviation)
Nitrogen Model			
Dairy	70.2 (56.9-92.5)	18.6 (10.4-27.6)	26.9 (9.3)
Non-Dairy Pasture	16.1 (14.2-19.5)	6.11 (3.82-8.02)	8.10 ^a (9.26)
Non-Pasture	5.05 (1.22-6.49)	3.20 (1.83-4.91)	2.35 (2.38)
Phosphorus Model			
Dairy Pasture	7.09 (6.36-8.19)	1.92 (1.26-2.54)	1.0 (0.29)
Other Pasture	3.71 (2.93-5.01)	1.17 (0.83-1.71)	0.60 ^a (1.0)
Non-Pasture	1.47 (0.76-3.17)	0.97 (0.45-2.31)	0.23 (0.32)

^aValue for hill pasture

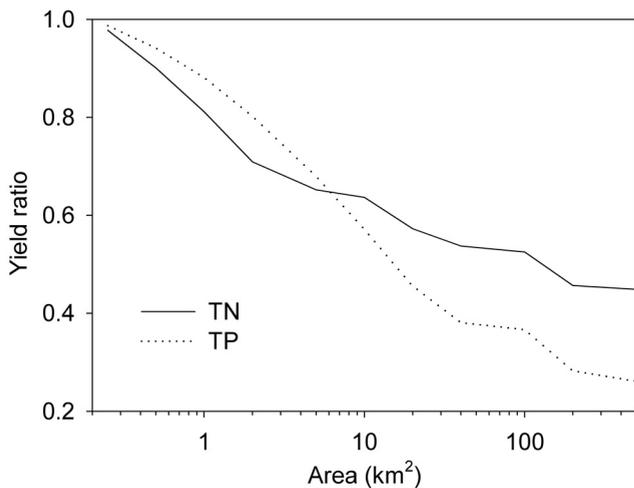


Figure 5 – Fraction of load entering streams which is delivered to the catchment outlet for catchments of various sizes (median values).

previous studies in New Zealand (Table 7). Attenuation does not reduce the yields for the non-pasture catchments as much as for pasture catchments, presumably because streams in the non-pasture catchments have higher flows and hence smaller in-stream attenuation coefficients.

The yields entering streams in SPARROW are greater than would be expected based on measurements of leaching in New Zealand, particularly for dairying (median predicted values of 70.2, 16.1, and 5.8 kg ha⁻¹ yr⁻¹ compared with measurements of 13, 14 and 4 kg ha⁻¹ yr⁻¹ respectively for dairy areas with less than 200 kg ha⁻¹ yr⁻¹ nitrogen fertiliser, sheep grazing areas with less than 50 kg ha⁻¹ yr⁻¹ fertiliser, and pine or bush catchments, with the measured values based on an unpublished review for Environment BOP). This discrepancy is compounded by the expectation that in pastoral settings we would expect some riparian attenuation of leached nutrients (Elliott and Sorrell, 2002). The discrepancy could be due to bias in the measurements of leaching, nutrient sources other than leaching (such as overland flow or dairy shed wastes) increasing the nutrient

load significantly beyond that for leaching alone, or overestimation by SPARROW of loads entering streams.

In the application of SPARROW to the Waikato (Alexander *et al.*, 2002), there were only two land-use classes, pasture and non-pasture, with yield coefficients of 49.6 kg ha⁻¹ yr⁻¹ and 5.97 kg ha⁻¹ yr⁻¹ respectively. The yield for pasture in the Waikato study is between that for dairy and other pasture in the present study. The yield for non-pasture is comparable to the yield for trees in the present study and greater than the yield for other non-pasture. Thus the two sets

of coefficients are broadly comparable. For further comparison with the Waikato study, the national model was calibrated using the same model form as for the Waikato (two land uses and a single flow cut-off at 1 m³ s⁻¹). This gave yield coefficients of 33 kg ha⁻¹ yr⁻¹ for pasture and 5.9 kg ha⁻¹ yr⁻¹ for non-pasture. The pasture value is lower than for the Waikato, which may reflect the greater proportion of dairy pasture in the Waikato compared with nationally. The RMSE for this simplified model (0.58) was considerably greater than for the Waikato model (0.38).

The source coefficients decrease in the order dairy>other pasture>trees>other non-pasture. This order is consistent with existing summaries of the relative yields for different land uses (Table 7), except that previous studies do not address the difference between trees and other non-pasture. The class other non-pasture land use, which covers 22.1% of the country, includes tussock, bare ground, horticultural areas, urban areas, water, and a number of other minor land use classes, but is dominated by tussock (13.6%) and bare ground (5.3%), which are associated with

low soil-fertility areas in New Zealand high-country or mountains. The yield for the other non-pasture land use is much lower than for trees, which reflects the predominance of tussock and bare ground in this class. While urban areas and horticultural areas could be expected to have higher yields (Painter *et al.*, 1997; Williamson, 1993) than tussock or bare ground, the model was not able to discern such differences, as the monitoring stations have only a small proportion of these land uses (<5.9% urban, <0.1% for horticulture).

The model was not able to distinguish differences in yield between beef, deer and sheep pastoral land uses. There is only a small fraction of deer land use in the catchments of the monitoring stations (<6.7%), so the model would not be expected to discern an effect of deer unless the yields were dramatically different from those of other land uses. We investigated beef as a separate source term. While the resulting source coefficient ($28 \text{ kg ha}^{-1} \text{ yr}^{-1}$) was larger than for other non-dairy pasture, there was high uncertainty associated with the coefficient (despite the high proportion of beef land use for some monitoring stations) so that it was not statistically different from the other coefficient. Further, including the beef term did not improve the R^2 value. Hence a separate term for beef was not retained. Similarly, the source coefficients for native bush, scrub, and exotic plantation forest, were not distinct so these land uses were lumped together.

The point-source coefficient (1.38) was greater than the value of 1.0 that would be expected if no adjustment were made to the point-source load, but there is high uncertainty associated with the coefficient and the confidence interval contains the value 1.0. The high uncertainty reflects the fact that the point-source loads inputs to the stream network above a monitoring station generally represent only a small fraction of the load at the station. The largest contribution

is for the Tarawera@Awakaponga site, where inputs from point sources are 38% of the load. This site was influential on the point-source coefficient (removing the site increased the variability of the coefficient and altering the point-source input upstream changed the value of the coefficient). A single point source was influential at this monitoring station, yet the load for that point source was not established tightly, as the effluent nutrients are not monitored closely and the treatment was improving over time. Changing the load for the site from 500 kg day^{-1} to 930 kg day^{-1} , an equally plausible value, reduced the point-source coefficient to 0.58. We also tried a model where separate source coefficients were applied for land versus water disposal of the waste, but the resulting coefficients were similar for both source types and within the large error bounds, so we were unable to distinguish whether land disposal led to a smaller delivery coefficient.

The exponent of the drainage term was -0.238 , so that the source of nitrogen decreases by 21% for each increase in drainage class, as the soil becomes better-drained. A similar trend was found in the Waikato application of SPARROW. Prior to this modelling, our expectation was that poorly-drained soils may have less nitrogen leaching, as water-logged soils are conducive to denitrification, but this has been contradicted by the modelling results. Potential reasons for increased nitrogen loads for poorly drained soils include increased solute bypass through clay soils, installation of artificial drains in poorly-drained soils leading to bypassing of riparian denitrification zones and deep subsurface denitrification zones, or more generation of overland flow leading to increased erosion of nutrient-enriched surface soils.

The exponent of the rain term was 0.243, so the load of nitrogen increases by 22% for each 1 m increase in mean annual rainfall. This is not surprising, as increased rainfall is expected to be associated with increased

erosion of particulate nitrogen in overland flow and leaching of mobile nitrogen through the soil profile.

The yield of nitrogen entering the streams (load entering a stream reach divided by the incremental catchment associated with the reach) is mapped in Figure 6. The yields are greatest in pasture areas (a large part of the North Island and the southeast part of the South Island), and are particularly high in areas where dairying is the predominant land use.

The total load of nitrogen entering the streams is 373,900 t yr⁻¹. This total load is

greater than the 132,200 t yr⁻¹ estimated to enter surface waters in New Zealand in the New Zealand State of the Environment Report (Taylor *et al.*, 1997), which was based on inventories of point-source loads and typical land-use yields (Hickey and Rutherford, 1986; Wilcock, 1986). Part of the discrepancy is due to the typical land-use yields being based on measurements of outputs from catchments typically >10 km² in area, so that they incorporate some in-stream attenuation, whereas the SPARROW estimates are without in-stream attenuation. Assuming the measurements correspond

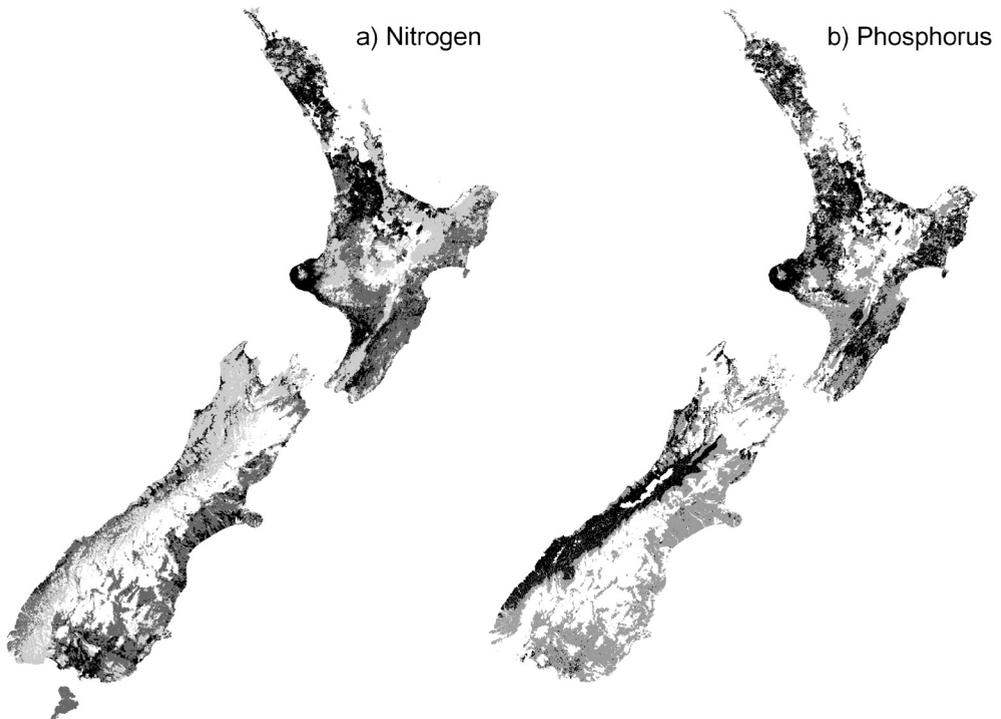


Figure 6 – Yields entering streams (load entering the stream reach divided by the area of the incremental catchment associated with the stream), for a) total nitrogen (TN) and b) total phosphorus (TP). Ranges for nitrogen are 0-5 (white), 5-10, 10-20, and >20 (black) kg ha⁻¹ yr⁻¹. Ranges for phosphorus are 0-2.5 (white), 2.5-5, and >5 (black) kg ha⁻¹ yr⁻¹.

Table 8 – Breakdown of land area and total nitrogen (TN) load by source. The loads exclude point sources discharging directly to the coast.

Source Type	Load Entering Streams	Load to Coast	Land use Area
Fraction of Total			
Point Source	1.8%	3.2%	–
Dairy	37.8%	36.7%	6.8%
Trees	19.7%	24.8%	39.2%
Other Pasture	38.9%	33.3%	31.9%
Other Non-pasture	1.8%	2.1%	22.1%
Total (t yr ⁻¹ or km ²)	373,900	167,700	263,500

typically to a 20 km² catchment and applying 1.8-fold increase in the non-point component of the load (Fig. 5) the corrected measurement-based load entering the streams would be more like 236,100 t yr⁻¹, which is still substantially less than the SPARROW value.

The breakdown of sources entering the streams by source type is shown in Table 8. Dairy land use makes a contribution to the load that is disproportionately large in relation to the area, while the other non-pasture areas make a disproportionately small contribution to the total load.

The load from point sources (6700 t yr⁻¹ after the source coefficient is applied) is smaller than that estimated previously (10,200 t yr⁻¹ Taylor *et al.*, 1997), and the relative contribution of point sources (1.8%) is less than estimated previously (7.7%). This is consistent with earlier observations of the changing role of point sources over time (Hickey and Rutherford, 1986).

In-stream and reservoir attenuation, and delivery to the coast

The in-stream decay coefficients are shown as a function of mean stream flow in Figure 7. As in earlier SPARROW studies (McMahon *et al.*, 2003) the stream decay coefficient (loss per unit stream length) increased as the flow decreased. Considering that stream dimensions (depth, width) are related to

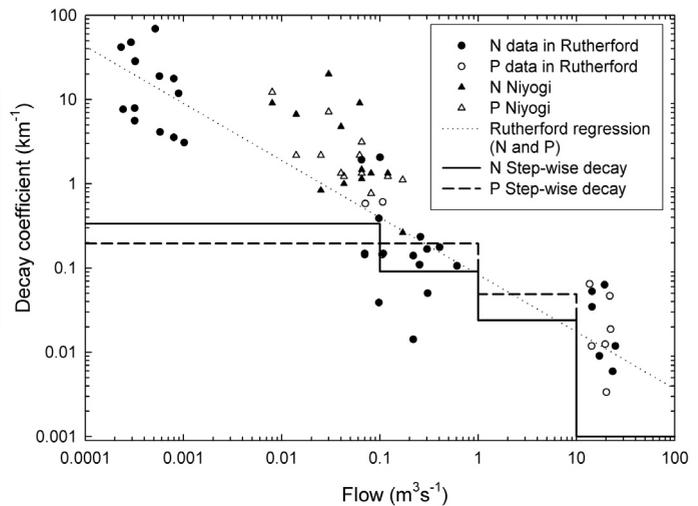


Figure 7 – Decay coefficients as a function of flow for the two models, and for various data from experiments in New Zealand (Niyogi *et al.*, 2004; Rutherford *et al.*, 1987), and a regression fitted to New Zealand data (Rutherford *et al.*, 1987; Cooper and Bottcher, 1993).

stream flow (Jowett, 1998) the higher decay coefficients also relate to shallower depths of flow and larger wetter perimeter per unit area of flow cross-section. Stream decay coefficients in other SPARROW models (e.g., Alexander *et al.*, 2000; Preston and Brakebill, 1999; Smith *et al.*, 1997) also show declines in the estimated rate of loss with increasing stream flow and channel depth. The inverse relation between the rate of nutrient loss and streamflow is consistent with current understanding of the biogeochemical processes responsible for the in-stream removal of nutrients (e.g., Alexander *et al.*, 2000). Streamflow and channel depth generally influence the extent of water and nutrient contact with benthic sediments and hyporheic zones—i.e., with increasing flow and water depth there is less water-sediment contact and thus less nutrient removal via denitrification and particulate storage.

The model decay coefficients are generally smaller than the experimental values (Fig. 8). The discrepancy is not surprising given the differences in time-scales and nutrient forms relating to the two sets of coefficients. The experimental values are usually determined by introducing soluble tracers and observing their uptake over a short time-scale (<1 day). In contrast, the SPARROW values relate to the total nutrients and the long-term (>annual) time-scales. Soluble nutrients are more amenable to biological uptake than are particulates, and nutrients trapped in the short term may be re-mobilised by later storms (e.g., Cooke, 1988). Both these effects would lead to lower decay coefficients compared with those from short-term soluble-nutrient experiments.

While the decay coefficient increases as the flow goes below $0.01 \text{ m}^3 \text{ s}^{-1}$ for the experimental data, it remains constant for the model. When another smaller flow class of $<0.01 \text{ m}^3 \text{ s}^{-1}$ was introduced, the resulting decay coefficient was similar to the coefficient for the next-larger

flow class (between 0.01 and $0.1 \text{ m}^3 \text{ s}^{-1}$). High variability was associated with these coefficients, though (standard error approximately 0.4 m^{-1}), so the model and associated monitoring data do not contain enough information to define the decay coefficients well for small flows.

There is some interaction between the source and decay coefficients, so a decrease in the sources can be compensated by a decrease in the decay, resulting in similar overall loads. This interaction was investigated by forcing the diffuse source coefficients to be halved, fixing the land-to-water delivery term, and then recalibrating the model. The recalibrated decay coefficients were a factor of approximately 2 to 4 smaller than with the original larger source coefficients, and the R^2 decreased from 0.956 to 0.933. The interaction is also illustrated by comparing the step-wise and continuous decay models. The sources coefficients are typically 50% larger in the step-wise model, and the total load entering the stream system is 57% larger, but this is offset by larger decay coefficients in the small streams ($<10 \text{ km}^2$), so that for catchments 15 km^2 or larger the load in the streams are within 5% for the two decay model forms (based on mean values). Consequently, predicted loads for streams with a catchment area less than 10 km^2 probably have much larger uncertainties than are reflected by the reported prediction errors and should be used with caution. If more observations were available for small streams (the smallest catchment in the existing monitoring station network is 13.7 km^2), then there would be more information to set the decay coefficients for smaller streams, the parameter interaction would likely be reduced, and the uncertainty associated with predicted loads for smaller streams would be reduced.

The estimated reservoir net settling velocity was 12.6 m yr^{-1} , which is larger than the coefficient of 4.38 m yr^{-1} in the earlier



Figure 8 – Fraction of total nitrogen (TN) load delivered to the coast from the outlet of the stream reach. Ranges are 0-0.25 (white), 0.25-0.75(grey), 0.75-1 (black).

Waikato application, but similar to that observed in a regional SPARROW model in the United States (McMahon *et al.*, 2003). Settling velocity coefficients of more than about 25 m yr^{-1} have typically been observed in lakes where algal uptake and particulate settling is a predominant process controlling nitrogen removal (Alexander *et al.*, 2002). Lower settling velocity coefficients, ranging

from 1 to nearly 20 m yr^{-1} , are more typical of lakes where denitrification has been reported to be the primary removal process (Alexander *et al.*, 2002).

A map of the fraction of load delivered to the coast from the outlet of stream reaches is shown in Figure 8. The delivery fraction is largest for areas with high water yields (dark areas along the west coast of the South Island) or for reaches feeding to large streams (darker lines along the larger streams), and is lowest for low-rainfall areas and areas distant from the main streams. This has implications in terms of managing nutrient loads to the coast or estuaries, as in some areas the sources are reduced by a factor of 4 or more by natural in-stream or reservoir attenuation processes (delivery coefficients less than 25%) while in other areas there is minimal attenuation.

The total nitrogen load reaching the coast is $167,700 \text{ t yr}^{-1}$ (apart from point-source discharges direct to coastal waters). The exported yield for the whole country is then $6.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$, which is similar to the mean value of the measurement stations ($5.3 \text{ kg ha}^{-1} \text{ yr}^{-1}$). The load to the coast is greater than the load estimated previously ($132,200 \text{ t yr}^{-1}$ —Taylor *et al.*, 1997). That estimate was based on inventories of point-source loads and typical land-use yields, so

after in-stream and reservoir attenuation the load would be less than 132,200 t yr⁻¹. This suggests that the load to the coast may have increased over time, but more likely just reflects differences in methods for estimating loads. Some 45% of the inputs to the streams are delivered to the coast.

The load attenuated in reservoirs is 7100 t yr⁻¹. Reservoir attenuation thus represents only 3.5% of the total loss between the sources entering the streams and the coast, the remainder being attributable to in-stream attenuation.

The relative contribution of different source types to the total nitrogen load delivered to the coast (Table 8) is similar to the breakdown of sources into the streams. The only exception is point sources, where the contribution to the coast is 3.2% rather than 1.8%, which results from the point sources contributing predominantly to larger streams that have less attenuation.

We found that accounting for nitrogen removal in streams and reservoirs, together with the use of a spatially referenced structure, reduced the overall model error. When the stream and reservoir decay coefficients were set to zero and the model was re-calibrated, R² reduced from 0.956 to 0.920, and the RMSE increased from 0.33 to 0.44. The source coefficients reduced considerably to compensate, and the rain delivery coefficient increased to 0.52 m⁻¹. When the land-to-water delivery coefficient was set to zero as well, R² reduced to 0.890, and the resulting source coefficients were 38.5, 3.7, 3.1, and 0.3 kg ha⁻¹ yr⁻¹ for dairy pasture, tree, other pasture, and other non-pasture land uses respectively. We also conducted a conventional regional regression, whereby the load for a reach was expressed as a linear combination of the sources in the catchment upstream, with log-transformation of the variables, removal of the Hakataramea outlier, and no land-to-water deliver terms. This resulted in an R² of 0.869 and similar

source coefficients. The difference between the conventional regional regression and the reduced SPARROW model arises because the SPARROW load is re-set at the measured value at monitoring stations during load accumulation.

Discussion of the phosphorus model

The performance of the total phosphorus model (R² of 0.900 and RMSE of 0.58) is substantially worse than for total nitrogen. The RMSE of the model is larger than the mean standard error of the measured loads (0.47), suggesting that there is room for improvement in the model before limitations associated with measurements come into play.

The yields of total phosphorus entering the streams for the various land uses (dairy pasture, other pasture, non-pasture) are reduced considerably through attenuation (Table 7 and Fig. 4). Even so, the yields for catchments between 20 and 30 km² are still larger than measured yields exported from catchments reported in previous New Zealand studies (Table 7). The yields from SPARROW decrease in the order dairy>other pasture>non-pasture, which is consistent with previous measurements. The point-source coefficient was 1.85, but had a high associated error. As with the nitrogen model, this coefficient is influenced strongly by a few sources.

The erosion source term, based on the Hicks and Shankar (2003) erosion yield maps, was significant. The significance of the term arises from the high observed phosphorus loads in some areas with a naturally-high sediment yield. The source coefficient for the erosion term is 0.255 g kg⁻¹, which can be considered as the concentration of phosphorus on the eroded sediment. This concentration is in the medium range of

acid-soluble phosphorus content of soil parent material based on the New Zealand Soils Database (Leathwick, 2002). For the mean erosion of $8.60 \text{ t ha}^{-1} \text{ yr}^{-1}$ in New Zealand, the yield of total phosphorus at the calibrated concentration is $2.2 \text{ kg ha}^{-1} \text{ yr}^{-1}$, which is considerable. We also attempted to use the erosion severity index instead of the sediment yield map to represent erosion, but the coefficient for the erosion index was not statistically significant and this term did not improve the model.

Our use of the Hicks and Shankar sediment yields as non-attenuated yields is somewhat inconsistent with the way in which those yields were derived. In the SPARROW model, the sediment source is attenuated in the same way as other sources, so the erosion yields are used as a non-attenuated yield. On the other hand, the map is based on calibration to measured loads in streams and hence it already includes some stream attenuation. The inconsistency is probably masked to some degree by allowing the concentration of total phosphorus on the sediment to be a calibrated model parameter. It would be desirable to resolve this inconsistency in the future.

The mean annual rainfall was a significant land-to-water delivery term, which is not surprising, as soil erosion is expected to be dependent on rainfall. The rain coefficient implies an 88% increase in phosphorus load for each metre increase in rainfall. Other delivery terms were not significant—this was surprising as it was thought that the potential erosion class, drainage class, and slope would all be related to erosion. It would be desirable to investigate forms of the delivery factor other than the exponential form, considering that with this form the very high rainfall in some areas leads to high delivery factors.

The yield of total phosphorus entering the streams is mapped in Figure 6. The highest yields for total phosphorus are associated

with high erosion areas, so the maps for total phosphorus and total nitrogen look different. High yields are also associated with pasture land use in higher-rainfall areas.

The attenuation coefficients for total phosphorus were greater than for total nitrogen for flows greater than $0.1 \text{ m}^3 \text{ s}^{-1}$ (Fig. 7), and the median net attenuation was a little greater for total phosphorus than for total nitrogen for areas larger than 20 km^2 (Fig. 5). The attenuation coefficients are generally in accord with values measured in previous studies (Fig. 7) for flows $>0.1 \text{ m}^3 \text{ s}^{-1}$. For smaller flows, the model could not discern any variation in decay coefficients. As with total nitrogen, more observations of loads from smaller catchments could help resolve the decay coefficient for smaller flows.

The estimated reservoir net settling velocity was 132 m yr^{-1} . This value is large compared with values reported in other applications of SPARROW (including the application to the Waikato) and typical values of $5\text{-}20 \text{ m yr}^{-1}$ from other studies of nutrient retention in lakes (Chapra, 1997). However, Chapra reports settling velocities up to 200 m yr^{-1} . No particular reason for the high settling velocity, such as load observations with a high influence on the settling velocity, could be found. The settling velocity was still high when the erosion source term was removed, so the high settling velocity is not related to the use of the erosion source term.

The predicted total phosphorus load delivered to the coast (excluding point sources discharging directly to the coast) is about $63,100 \text{ t yr}^{-1}$ (Table 9). The implied yield for the country is $2.4 \text{ kg ha yr}^{-1}$, which is greater than the mean of the measured load at the monitoring stations ($1.28 \text{ kg ha yr}^{-1}$). Based on Rutherford *et al.* (1987), the load is approximately $20,000 \text{ t/year}$ (based on compilations of point sources and measured yields for a range of land uses), which is considerably less than the SPARROW

Table 9 – Breakdown of land area and total phosphorus (TP) load by source type. The loads exclude point sources discharging directly to the coast.

Source Type	Load Entering Streams	Load to Coast	Land use Area
Fraction of Total			
Point Source	1.2%	1.8%	–
Dairy	9.9%	8.4%	6.8%
Other Pasture	21.9%	17.0%	31.9%
Non-pasture	17.7%	19.5%	61.2%
Sediment	49.3%	53.2%	-
Total (t yr ⁻¹ or km ²)	143,403	63,057	263,500

estimate. Some 53.2 % of the predicted load is associated with the erosion source in the SPARROW model (Table 9). As the erosion source is added to the land-use-specific sources, it is difficult with this particular model specification to quantify the separate contributions of the various land uses

Point sources, excluding discharges directly to the coast, are predicted to contribute only 1.8% of the load to the coast, or 1134 t yr⁻¹. Point-source inputs to streams amount to 1704 t yr⁻¹ (921 t yr⁻¹ before the delivery coefficient is applied), which is less than the 2230 t yr⁻¹ estimated by Rutherford *et al.* (1987).

The model predicts that the load to the coast is 44% of the load entering the streams. The attenuation is dominated by stream attenuation, with reservoir attenuation accounting for 8.3% of the overall attenuation.

Management implications and future work

The key land-based sources of nitrogen to the coast of New Zealand are pastoral land uses, especially dairy land use, while point sources are a relatively minor issue. The quality of lake and stream waters is of more concern than discharges to the coast in New Zealand, but the same key nitrogen sources apply. An

exception is downstream of the largest point sources, such as on the lower Waikato and Tarawera rivers. As dairying is increasing in New Zealand, it seems likely that the loads of nitrogen into streams and lakes will rise in the future. Pastoral land uses also make a high contribution to total phosphorus loads, although there is a high natural component of phosphorus loads associated with erosion in some areas.

The final SPARROW model does not include pasture management variables such as amounts of fertiliser applied, stocking density, grazing management, or riparian retirement. Fertiliser use is increasing (Parliamentary Commissioner for the Environment, 2004), so the model could under-predict the load of nutrients in the future if the model is not re-calibrated. Further, the model is not at present able to predict the effects of modifying source controls such as pasture management, and is limited to predicting the effects of complete changes in land use or the removal of point sources. At present we do not have suitable databases for most of these management variables, although Agribase does contain information on stocking density. An alternative approach is to use other models (such as soil leaching models) or farm-scale measurements to derive load adjustment factors, which could then be applied to the SPARROW source predictions.

The SPARROW models here are calibrated to mean annual stream nutrient conditions for 1996-1999, and do not take account of time lags between past changes in land use and the resulting changes in loading to streams. In some locations in New Zealand the concentration of nitrogen in streams is known to be rising following conversion to pasture decades ago, which is related in part to the time for water to move through the groundwater and into streams (Vant and Smith, 2002). If the monitoring stations used in the calibration are subject to such lags, there is a danger that SPARROW could under-estimate the sources of nitrogen in the future. From trend analysis of the national monitoring data over the period 1989-1998 (Scarsbrook *et al.*, 2003), concentrations of nitrogen are increasing faster at non-baseline sites than at baseline sites (that is, sites with little land development in the catchment). However, we do not know whether this is due to the lagged effect of past changes in land use or whether it is due to ongoing changes in land use. We cannot assess the severity of this lag effect without assessing changes in land use. There would be considerable value in incorporating the information from an assessment of changes in land use together with historical stream monitoring data into a temporal SPARROW model.

It would be desirable to add more measured load data from small streams to the model, to allow the attenuation coefficient and source terms to be evaluated with more certainty. At present, changes in the source coefficients can be offset to a large degree by changes in the decay coefficient without affecting the overall model predictions for larger streams. Inclusion of monitored loads from regional monitoring programmes may not provide more data from smaller streams. For example, the smallest catchment area for the set of Waikato regional monitoring sites is 22 km². Inclusion of such regional monitoring data could be useful for refining other aspects of

the model, though, and for validation of the model.

Another potential limitation of the final form of the model is that the in-stream attenuation is independent of the degree of stream shading or type of in-stream vegetation, yet studies in New Zealand show that these factors can affect in-stream attenuation rates (Cooper and Thomson, 1988; Howard-Williams and Pickmere, 1999; Wilcock *et al.*, 2002). While it would be desirable to incorporate these effects into the model, there are no suitable databases on riparian conditions to provide input to the model. Also, there is currently a poor understanding of how well these measured uptake rates, which typically describe the short-term, temporary uptake of nutrients in streams by aquatic vegetation, reflect the long-term multi-year processing and storage of nutrients in waterways, which the current SPARROW calibrations tend to mirror. Additional experimental research is needed over longer periods to improve understanding of the annual cycling and fate of nutrients in New Zealand streams and other streams in temperate ecosystems.

The calibrated model is being disseminated in two forms. First, the predicted nitrogen loads have been made available on the internet through a NIWA web-based interactive map system, Freshwater Information New Zealand (FINZ). Also, the model is being incorporated into a desktop GIS application to allow the implications of changes in land use on nutrient loads to be determined for a river basin selected by the user.

Summary and conclusions

The model SPARROW was used successfully to model the mean annual load of nitrogen (TN) and phosphorus (TP) for New Zealand (area 270,000 km²). The model was

calibrated to the 77 stations in the national water quality monitoring network.

For total nitrogen, the model explained about 95.6% of the spatial variability in the log-transformed estimates of stream load (R^2 of 0.956). The RMSE was 0.33 in natural-log space (39% standard error in non-transformed space). In comparison, a conventional regional regression without stream and reservoir attenuation and the associated spatial framework explained only 87% of the spatial variability in stream loads. The performance for the total phosphorus model was worse than for total nitrogen, giving an R^2 of 0.900 and RMSE of 0.58 (79% error).

The model distinguished between four diffuse-source types for total nitrogen: dairy pasture, non-dairy pasture, trees, and other non-pasture. Further refinement of the land-use categories did not improve the model, while further aggregation of the land uses degraded the model. The soil internal drainage class and mean annual rainfall were used as a land-to-water delivery factors for total nitrogen. For total phosphorus the non-pasture land uses were lumped together, while a new source based on available maps of erosion was introduced. Rainfall was the only delivery factor retained in the model for total phosphorus.

The predictions of exported yields for streams with catchments $> 20 \text{ km}^2$ are broadly comparable with previous compilations of yields for various land use classes for nitrogen, but are larger than the previous measurements for phosphorus. The predicted relative ordering of the yields for the various land uses is consistent with the previous measurements. The yield coefficients for total nitrogen are broadly comparable with the previous application of SPARROW to the Waikato river basin in New Zealand. The point-source coefficient was greater than 1, but had a large associated error and

was influenced strongly by a single point source.

The predicted load of total nitrogen delivered to the coast was $167,700 \text{ t yr}^{-1}$, which is 45% of the loads entering the streams. For total phosphorus the predicted load to the coast was $63,100 \text{ t yr}^{-1}$, 44% of the load entering the streams. Reservoir/lake attenuation makes a relatively small contribution to the overall attenuation compared with in-stream attenuation (3.5 % for nitrogen and 8.5% for phosphorus). The largest contribution of total nitrogen is from pastoral land uses, together accounting for 70.0% of the total nitrogen load to the coast. Dairy land use makes a disproportionately large contribution to the load of total nitrogen in relation to the area of land (36.7% of the load versus 6.8% of the land). For total phosphorus the highest contribution of the load to the coast is from erosion (53.2%). Point sources contribute only a small proportion of the load to the coast (3.2% for nitrogen, 1.8% for total phosphorus, excluding point sources discharging directly into coastal waters).

The first order in-stream decay coefficients (expressed per unit length of stream) reduce as the flow rate increases, which is consistent with previous measurements of attenuation rates and previous applications of SPARROW. For total nitrogen the attenuation rates are smaller than measurements, which could be related to the relatively short time-scale of the measurements and their emphasis on soluble nutrients. The lake/reservoir effective settling velocity for total nitrogen (12.6 m yr^{-1}) was within the expected range. For total phosphorus the settling coefficient (132 m yr^{-1}) was higher than expected based on previous applications of SPARROW.

There was some interaction between the source coefficients and the decay coefficients for small streams, so an increase

in the attenuation in small streams could compensate to some degree for an increase in sources. The model for total nitrogen with the attenuation coefficient varying continuously as a power function of flow gave a model R^2 and RMSE fit close to that for a step-wise variation of decay. However, the source coefficients were lower for the continuous model, which was offset by lower decay coefficients. The predicted load was comparable for the two decay models only for streams with catchments $>20 \text{ km}^2$. These results, and the fact that the monitored loads were for catchments smaller than 14 km^2 , suggest that the model predictions of stream load for catchments less than $10\text{-}20 \text{ km}^2$ have large uncertainties and should be used with caution. Including smaller streams in the calibration dataset would likely give more reliable estimates of yields for the smaller catchments.

There is scope for further refinements to the model, including: improvement of the erosion source term for phosphorus; development of a temporal model to incorporate time lags between a changes in land use and changes in nutrient loads; inclusion of data from regional monitoring programmes to refine model parameters and provide model validation; and inclusion of land management practices.

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