Application of Physiographic-based modelling to estimate contaminant load to the Hokianga Harbour

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Executive Summary

Improving the health of the Hokianga Harbour and catchment is important to the Northland Regional Council and the local community. To understand the processes and land use activities influencing water quality, a physiographic approach has been applied to assess the current water quality state, and model current and pre-human nutrient (nitrogen and phosphorus), sediment and *E. coli* loads to the Hokianga harbour catchment. This report summaries the information for the Hokianga Harbour catchment utilising the results from a regional assessment of Northland's steady state water quality (Rissmann and Pearson, 2020) and catchment load estimates (Pearson et al., 2020).

Water Quality State

Water quality data collected over a 5-year period (2015-2019) was analysed to assess water quality state at six locations in the Hokianga catchment against the National Policy Statement for Freshwater Management National Objectives Framework (NOF; Ministry for the Environment 2020). State assessments are undertaken for ecosystem health (water quality) as indicated by nitrate and ammonia toxicity, dissolved reactive phosphorus, and suspended fine sediment (clarity), and human contact (recreational use) as indicated by *E.coli*. The regional steady-state model values and attribute state of Rissmann and Pearson is provided for comparison with the measured data for all attributes.

Water quality state in the Hokianga catchment indicates that for nitrate toxicity all State of Environment monitoring sites are above national bottom lines (A band for both median and 95th percentile). For ammonia toxicity annual medians are in the A band while one site fails the national bottom line during the annual maximum (C band).

There are three sites in the Hokianga harbour catchment that are in the D band for dissolved reactive phosphorus, including two sites that are predominantly natural state. Sites can be classified in the C or D band despite being predominantly natural state due to the underlying geology. Basaltic rocks commonly contain higher elemental phosphorus concentrations than felsic sedimentary rock and weathers faster than felsic rock, supplying inorganic phosphorus to the river network. Although DRP may be elevated, on its own it isn't sufficient to favour eutrophication which requires elevated dissolved inorganic nitrogen for increase algal and plant growth.

The two lake-fed sites in the Hokianga harbour catchment fail the newly introduced national bottom line for suspended fine sediment as indicated by water clarity. Water clarity at these sites is similar to other soft bottom lowland sites in the catchment. However, the national bottom line is significantly higher for lake-fed rivers (>2.22m for lake fed compared to >0.61m for soft bottom lowland). Exemptions exist for naturally occurring processes which reduce visual clarity, such as autochthonous phytoplankton production in warm climates. It is important to note that while instream clarity may have a low to moderate impact on instream biota, the effect of sediment accumulation in the harbour is likey to be significantly more detrimental in the harbour resulting in a loss of biodiversity and ecological value.

All predominantly agricultural sites in the Hokianga harbour catchment fall into the D and E bands for human contact as indicated by *E.coli*. Overall, particulate transport of sediment remains a pervasive and important water quality issue for the catchment, many of the processes controlling these issues are similar for *E.coli*.

Catchment Load

To estimate load across the Hokianga harbour catchment, we used a 5-year time period (2015-2019) and applied thirteen of the most common river load algorithms to each of Northland's 31 continuous flow and water quality sites (Nava et al., 2019; Snelder et al., 2014, 2017; Pearson et al., 2020). The median load estimate at each site was area-weighted and modelled using the physiographic process attribute gradient classification and machine learning (Rissmann and Pearson, 2020; Pearson et al., 2020).

Significant land use change has occurred in the Hokianga harbour catchment with only 45% of the natural land cover remaining in the catchment. Diffuse anthropogenic sources, predominantly agricultural and forestry, make up the largest contribution of contaminant load to the Hokianga harbour. Natural state contaminant load is now a minor component of the total contaminant load, except for dissolved reactive phosphorus.

Hokianga harbour catchment budget in T/year for TN, NNN, TP, DRP, and TSS and CFU 10^9 for E.coli.

	Area (Ha)	TN	NNN	TP	DRP	TSS	*E.coli*
Diffuse sources							
Land in natural state	72,404.0	540.86	132.50	58.45	32.73	4,780.84	3,026,487
Agricultural and forestry land use	86,212.0	1,104.13	328.32	102.92	34.40	14,945.54	11,073,082
Diffuse Source Total		1,644.99	460.82	161.37	67.13	19,726.38	14,099,569
Point sources							
Wastewater treatment plants		27.02	5.38	3.31	2.04	43.61	77,682
Farm dairy effluent to water		5.18		1.31		11.72	1,365,178
Point Source Total		32.20	5.38	4.62	2.04	55.33	1,442,860
Total catchment load		**1,677.19**	**466.20**	**165.99**	**69.17**	**19,781.71**	**15,542,430**
% Load from land in natural state		32.25	28.42	35.21	47.31	24.17	19.47
% Load from agricultural land and anthropogenic sources		67.75	71.58	64.79	52.69	75.83	80.53

Note: Total catchment load is the sum of both the diffuse and point source discharges. NNN and DRP loads from farm dairy effluent were not calculated.

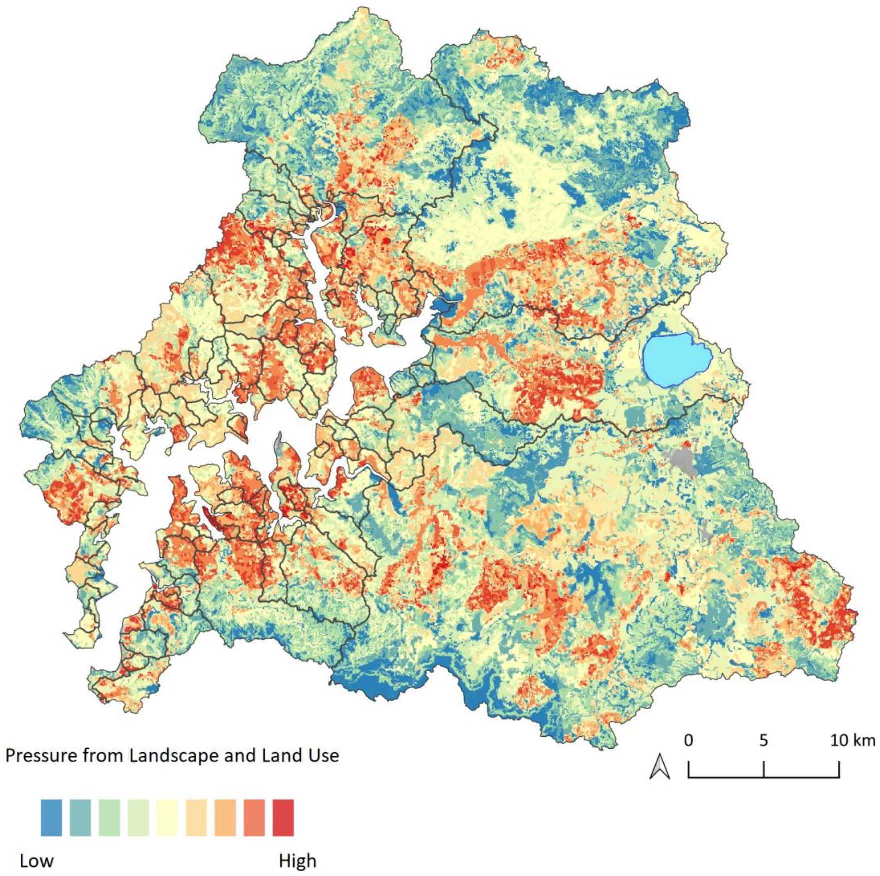
Total nitrogen (TN) load has increased approximately by 30% from pre-human load due to anthropogenic derived sources and now represents 68% of the total input. Due to the landscape characteristics, the dominant form of nitrogen is organic, which accounts for approximately two-thirds of the TN load. Total phosphorus (TP) load has increased by approximately 28% due to anthropogenic derived sources and now represents 65% of the total input. Weathering processes and sediment erosion contribute approximately half of the dissolved reactive phosphorus from predominantly natural state areas due to the high phosphorus concentent in the underlying Tangihua Basalt. However, the main form of phosphorus load to the harbour is from particulate phosphorus. With regards to sediment, the weak and highly erodible sedimentary rocks of the Northland Allochthon are particularly susceptible to sediment loss especially under intensive land uses. The sediment load to the harbour has nearly doubled (48%) since pre-human times, with approximately 76% now derived from anthropogenic sources. It is estimated that 81% of the microbial load is from diffuse agricultural sources; an increase of approximately 64% from the pre-human load estimate.

Pre-human and current load to the Hokianga Harbour. TN, TP, and TSS is in tonnes per year and microbial load as indicated by E.coli in CFU 10^9 per year.

	TN	TP	TSS	*E.coli*
Pre-human load	1,169.38	119.46	10,331.84	6,543,497
Current load	1,677.19	165.99	19,781.71	15,542,430
Anthropogenic increase over pre-human load (%)	30.3	28.0	47.8	57.9

Implications for catchment management

Physiographic science has shown that the landscape attributes characteristic of a catchment strongly influence the type and severity of water quality issues. The figure below shows where the combined risk to water quality from landscape (erosion) and land use pressures are in the Hokianga Harbour catchment. In addition to sediment, these areas identified as high risk (red) are also likely to be contributing a disproportionately high microbial load to the harbour as surficial runoff is a common occurrence in poorly-drained heavy clay soils. Landscape knowledge is vital for guiding investment in mitigations that are appropriately targeted and cost-effective but also for generating robust policy that is relevant to land users and their communities.



Combined landscape and land use risk to water quality. Where blue areas denote low landscape and land use risk to water quality, and red areas denoted a high combined landscape and land use risk. Mapped from the erosion susceptibility classification of Rissmann et al. (2018b) and the physiographic land intensity layer (ESC + LUI).

1 Introduction

Improving the health of the Hokianga Harbour and catchment is important to the Northland Regional Council and the local community. Under the National Policy Statement for Freshwater Management (NPS-FM; Ministry for the Environment, 2017, 2020) the council is required to set limits to ensure that freshwater objectives can be met. These discharge limits could be expressed as annual loads to the catchment. Through the allocation of a part of the annual catchment load to all land and water resource users in the upstream catchment, contaminants can be managed. Therefore, a fundamental component of policy development and monitoring progress towards catchment objectives is the estimation of catchment nutrient loads. To understand the processes and activities influencing water quality in the catchment and Harbour, the physiographic approach is applied to model current and pre-human nutrient (nitrogen and phosphorus), sediment and *E. coli* loads.

Catchment contaminant loads are estimated based on calculations that combine discrete contaminant concentration observations with more frequent observations of flow. There are numerous calculation methods for converting water quality and flow data into contaminant load estimates. A recent review of load methodologies in New Zealand by Snelder et al. (2017) concluded that "there is no single preferred load calculation method" and "regulatory authorities should be aware that the precision of loads estimated from monthly data is likely to be "optimistic" with respect to the actual repeatability of load estimates." Bearing these inherent limitations in mind, we use a 5-year time period (2015-2019) and apply 13 of the most common compound river load algorithms to each of Northland's 31 continuous flow and water quality sites. Median load estimates are then combined with Physiographic layers to model loads for unmonitored parts of the catchment.

This report summarises the information for the Hokianga Harbour catchment utilising the results from an assessment of Northland's steady state water quality (Rissmann and Pearson, 2020) and catchment load estimates (Pearson et al., 2020). Underpinning these reports is the physiographic application to the Northland region (Rissmann et al., 2018a) which includes high-resolution mapping of erosion susceptibility (Rissmann et al., 2018b) and wetness gradients (Rissmann et al., 2019b,c). During the initial physiographic mapping of the region, hydrochemical evaluation of regional ground and surface water was undertaken (Rissmann et al., 2018a). To put current load estimates into context for the Hokianga Harbour, loads were estimated for natural state (i.e. pre – human) condition to identify a natural 'base' load and anthropogenic contribution. Anthropogenic contributions accounted for both diffuse and point source discharges.

2 Hokianga Harbour Catchment

2.1 Catchment Geology and Hydrology

The Hokianga Harbour is a long, drowned river valley estuarine system on the west coast in the Far North District, Northland (Figure 1). The estuary extends inland for 30 kilometres from the Tasman Sea. The catchment area is approximately 160,000 ha, of which the Waima River is the largest subcatchment at 51,954 ha (33% of the catchment area). Rainfall ranges from mean annual averages of 1,200 mm in low lying coastal areas up to 1,800 mm at high elevations (Chappell, 2013). High intensity rainfall events (max. 47 mm/hr) associated with the passage of tropical or subtropical storms can lead to significant overland flow and surface flooding in low lying areas (Chappell, 2013).

The geology of the catchment is predominantly weak and highly erodible sedimentary rocks of the Northland Allochthon (QMAP; Isaac, 1996; Edbrooke and Brook, 2009). The Northland Allochthon is mostly comprised of marine clastics, mudstones and sandstones, including but not limited to the Punakitere Sandstone and Mudstones and tectonically derived melange (as well as volcanic piles of Tangihua Complex pillow basalts) (Figure 1). Weak sedimentary rocks, such as the Punakitere Sandstone and Mudstones, are over steepened as they ramp up over the steep fore slopes of the more competent Tangihua Complex Basalts (indicated by cross-hatching in Figure 1). Over steepening and vegetative clearance across the lower elevation sedimentary units favour instability and erosion (Rissmann et al., 2018b). Over steepening of weak sedimentary units is thought to occur in response to slow compression against the more competent Tangihua Complex Basalts.

To the east of the catchment, young flood basalts of the Kerikeri Volcanic Group have erupted through or flooded overtop of the weak sedimentary lithologies of the Northland Allochthon (Figure 1; QMAP; Isaac, 1996; Edbrooke and Brook, 2009). In this setting, erosion is driven along the margins of the flood basalt in response to knickpoint migration and under cliffing of more competent volcanic rocks that is evidenced by the embayment of the Kerikeri Volcanics in the east and Waipoua Complex Basalts (small extent to the south of the catchment). Stratigraphically controlled erosional features are readily apparent in the vicinity of the Waihou Valley where the Utakura and Waikaraka streams drain the Kerikeri Volcanic Plateau to the Hokianga Harbour (Rissmann et al., 2018b). Similarly, knickpoint driven structural collapse and embayment are a strong feature of the Waipoua Basalts generating sediment that is subsequently transported to the Hokianga Harbour via the Mangatawa and neighbouring streams.

The influence of the geology over water quality and contaminant load is an important control and is discussed further in later sections of this report.

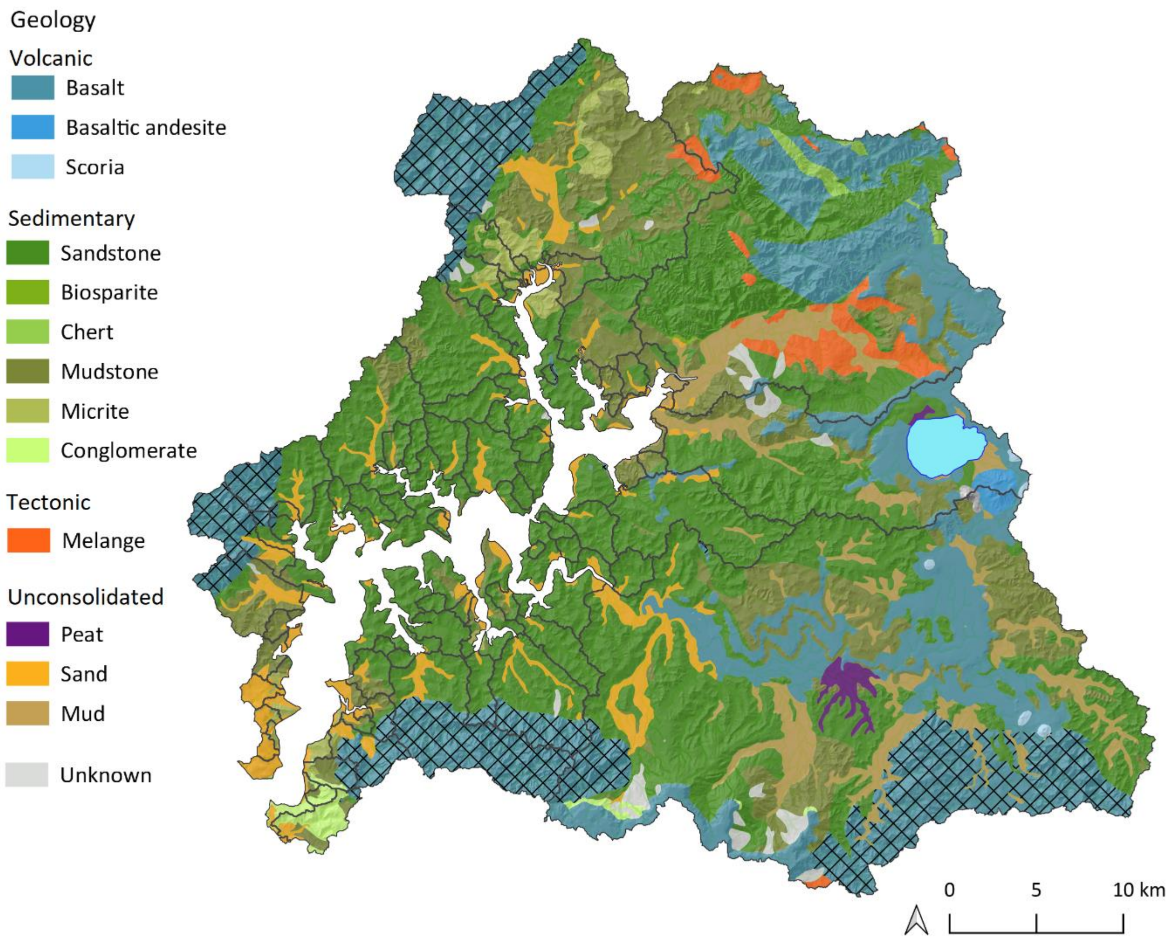


Figure 1. Main geology of the Hokianga Harbour Catchment. The area of Tangihua Basalt is indicated by a cross hatch. Data sourced from QMAP and symbolised from 'rock type' (Isaac, 1996; Edbrooke and Brook, 2009).

Northland Regional Council monitors water quality at six monitoring sites within the harbour catchment at: Punakaitere at Taheke; Utakura at Horeke Road; Utakura at Okaka Road; Waipapa at Forest Ranger; Mangamuka at Iwitaua Road; and Tapapa at SH1. Their locations and contributing catchment areas are shown in Figure 2. The river network and associated capture areas were derived from the River Environment Classification version 2.4 (Snelder and Biggs, 2002).

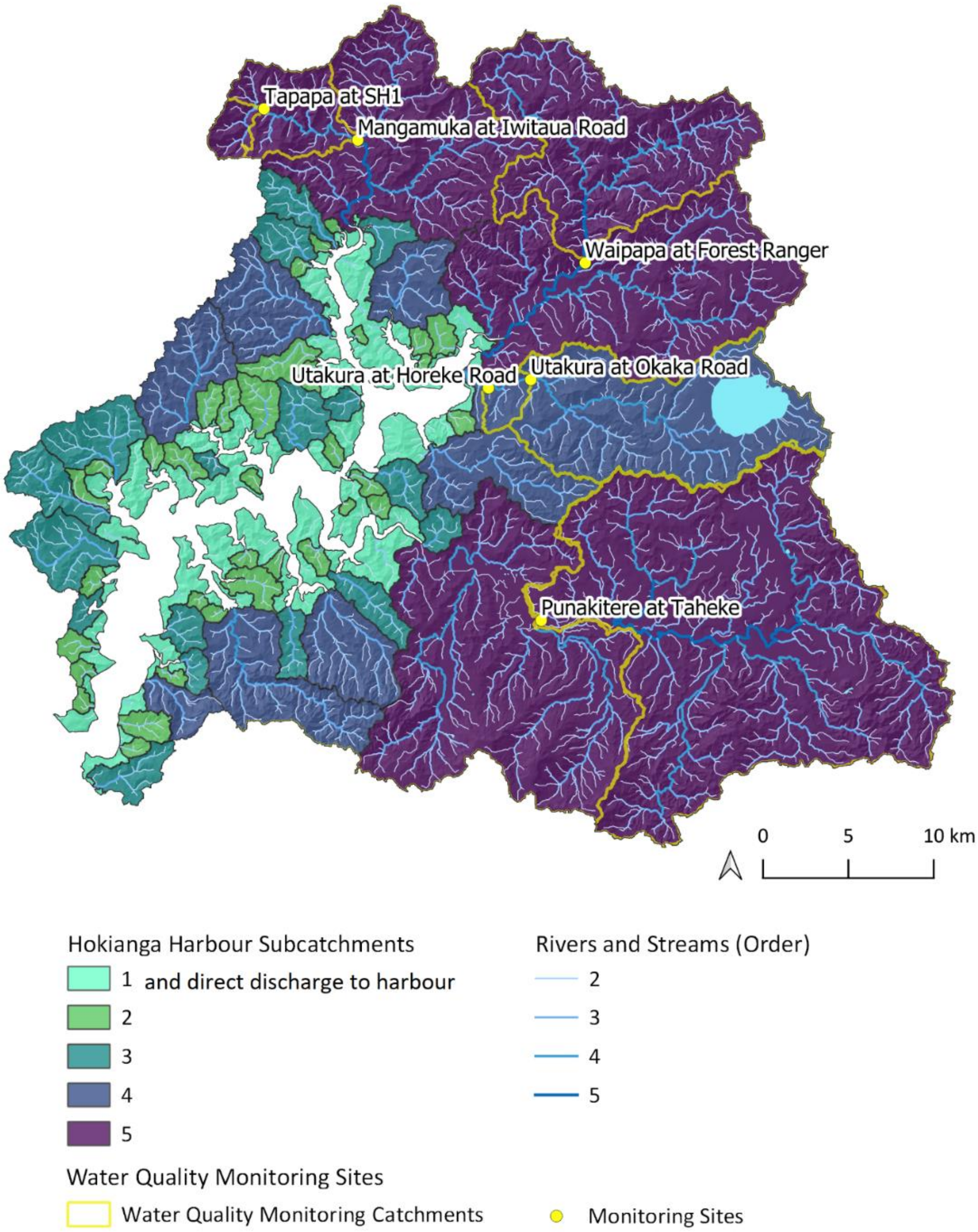


Figure 2. Hokianga Harbour riverlines and subcatchment boundaries (Data sourced from RECv2.4). Riverlines are shown for Order 2 streams and above. The zone of direct discharge and 1st order streams have been grouped together.

2.2 Land Use

The land use in the catchment is predominantly native forest (44%) followed by agricultural land (40%), and plantation forestry (13.5%) (Table 1, Figure 3; LUCAS, Ministry for the Environment, 2016). Of the agricultural land uses, drystock sheep and beef make up 33% of the catchment area, 6.9% is dairy, and 0.12% is cropland.

There are a number of point source discharges in the catchment, predominantly from Farm Dairy Effluent (FDE), stormwater, and wastewater treatment plants (WWTP; Figure 3, data provided by NRC). There are four WWTP in the catchment, three of which are near the harbour. The largest of the WWTP is Kaikohe, located in the upper Waima River catchment.

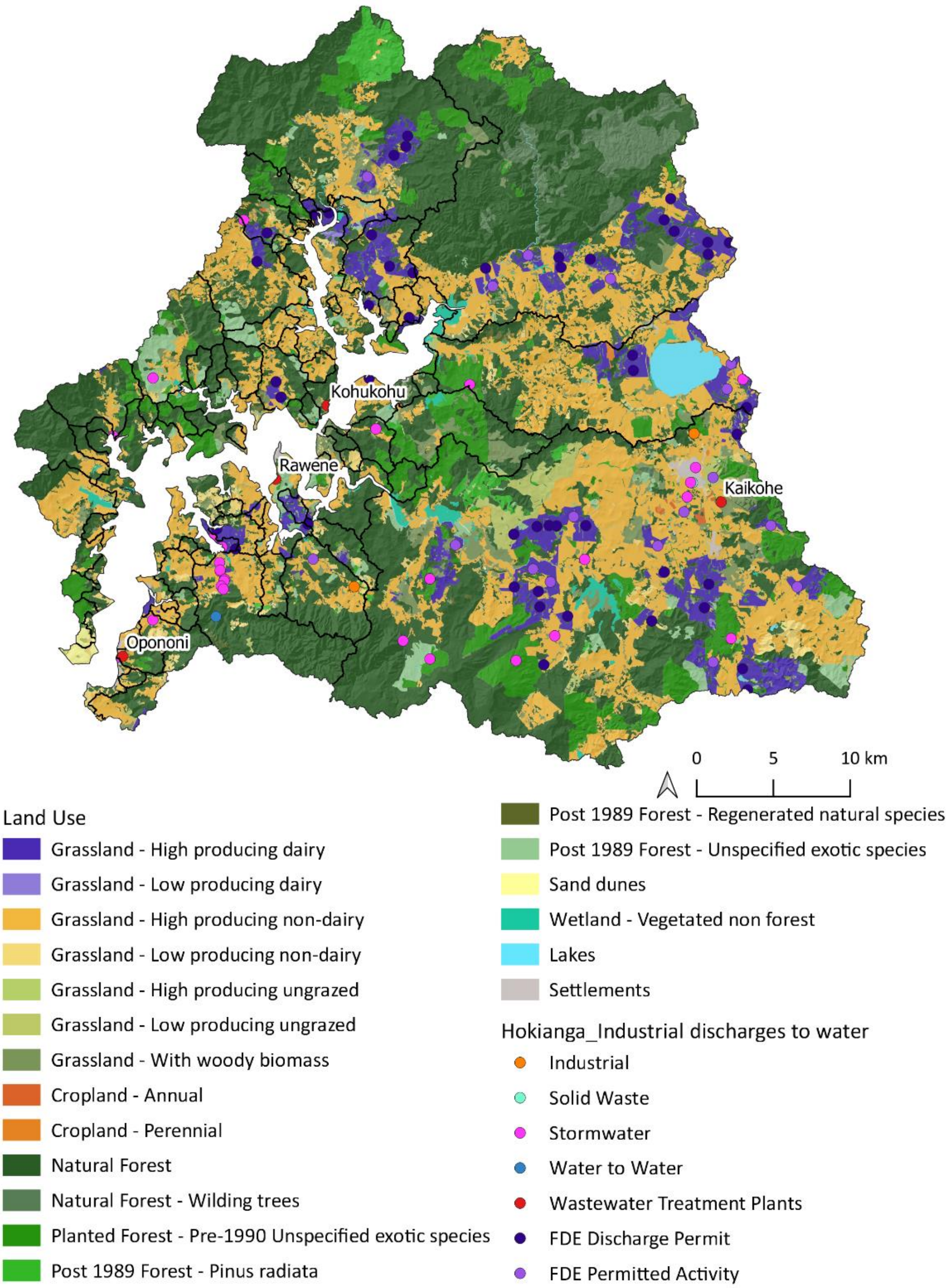


Figure 3. Land use in the Hokianga catchment (Data sourced from LUCAS, Ministry for the Environment, 2016). Industrial discharges are shown as points.

Table 1. Summary of land use in the Hokianga Harbour. Data sourced from Ministry for the Environment LUCAS land use map and Northland Regional Council data for point source discharges.

Land use	Area (ha)	Percentage of catchment
Native forest	70,541	44.06
Grassland – Dry stock	52,802	32.98
Plantation forest	21,620	13.51
Grassland - Dairy	11,016	6.88
Wetland	1,497	0.94
Water	1,472	0.92
Settlements	576	0.36
Other	366	0.23
Cropland	198	0.12

2.3 Pre-human Land Cover

Pre-human land cover for the Hokianga Harbour was obtained from the Potential Vegetation of New Zealand map (Manaaki Whenua Landcare Research, 2012; Figure 4). The potential forest composition was predicted from regressions relating the distributions of major canopy tree species to the environment. Environmental variables, chosen for their correspondence to major tree physiological processes, included annual and seasonal temperature and solar radiation, soil and atmospheric water deficit, soil leaching, slope, and soil parent material and drainage.

The land cover in the Hokianga was predominantly Kauri forest, some of which remains in the catchment today. Removal of the natural forest cover occured as early as 1820's as kauri logging became popular, and the land converted to pasture (Northland Regional Council, 2013). Areas where soils are poorly drained and water accumulates, such as in valley bottoms and surrounding lakes, species suited to wetland environments such as Kahikatea dominate (Figure 4).

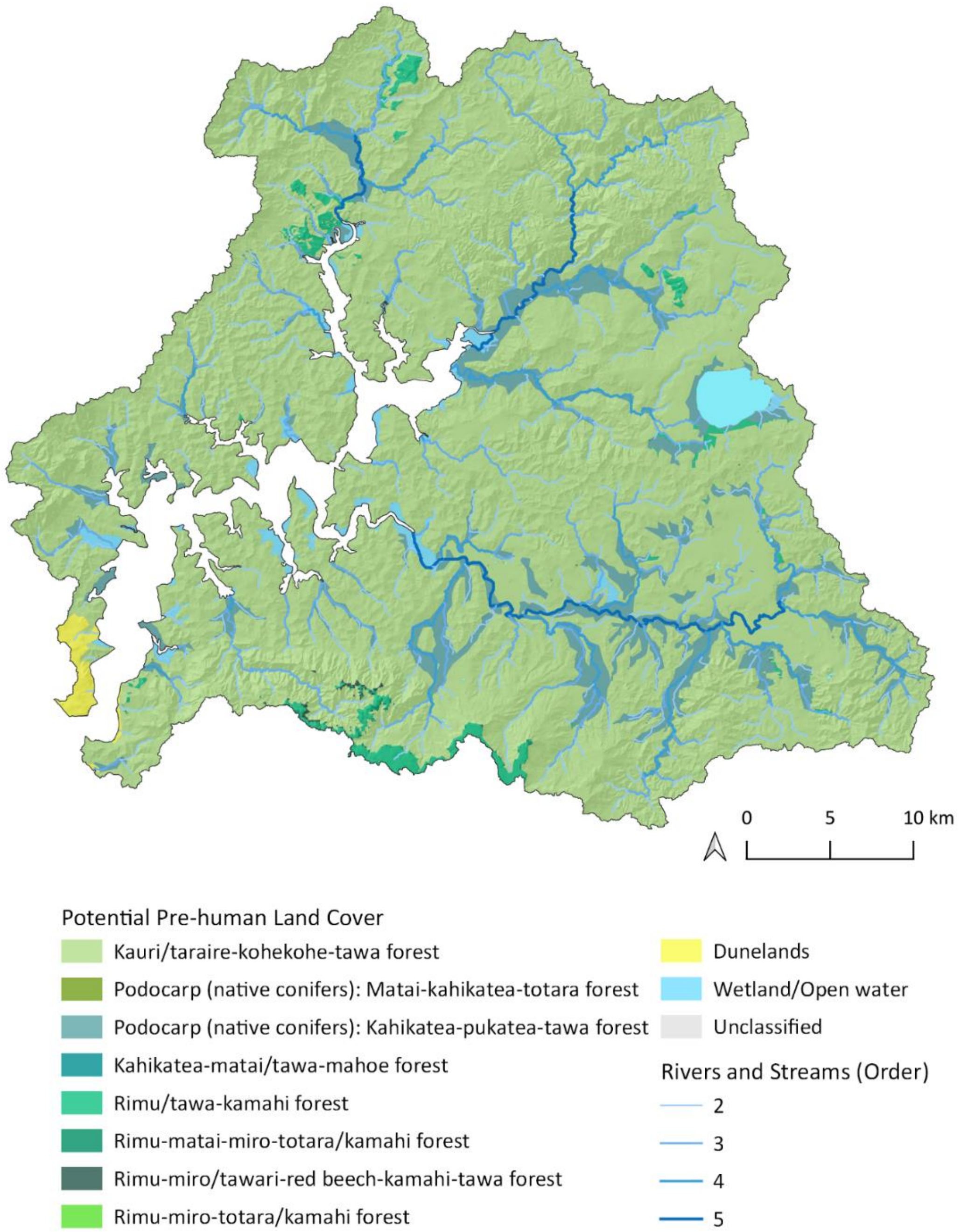


Figure 4. Pre-human land cover. Data sourced from Potential Vegetation of New Zealand, Manaaki Whenua Landcare Research, 2012.

3 Physiographic Approach for Water Quality Modelling and Loads

3.1 Physiographic Approach to Water Quality – Integrating Land and Water

Water quality varies in rivers and streams due to land use and variation in landscape characteristics. For example, the colour is different in water flowing from a wetland due to the high quantity of organic matter compared to water draining from a high-altitude hill-country catchment. Some waters are 'hard' due to an abundance of minerals, such as calcium and magnesium, whereas others are 'soft' because they contain only minor concentrations of these ions. Some waters contain high concentrations of sediment others do not. The reason for this variation in water composition is often due to the different characteristics of the natural landscape in addition to land use pressures.

Other landscape characteristics, such as soil type and topography, also greatly influence water quality. For example, overland flow or runoff across the land surface to waterways is more common where soils are slowly permeable and imperfectly to poorly drained (Figure 5). Where fine-textured and poorly drained soils dominate a farm or a catchment, the risk of runoff and associated sediment, phosphorus, and microbial loss to waterways is elevated. Where soils are permeable and well-drained, the risk of runoff occurring is lower. Here, most contaminants are removed (attenuated) during the deep percolation of water down through the soil. Although areas of well-drained soils are great at filtering out sediment, phosphorus, and microbes from water they do tend to be leakier to nitrate. Whether or not the nitrate builds up in the aquifer underlying well-drained soils will depend on the characteristics of the underlying aquifer into which it drains. Specifically, if the aquifer is comprised of materials that favour the natural removal of nitrate (denitrification), then leached nitrate is likely to be removed before reaching the stream. Groundwater hydrochemical measures are used to identify the types of rock and sediment that naturally favour the removal of nitrate within an aquifer.

Surface water hydrochemistry is used to identify the likely water source by evaluating the chemical fingerprint of the water against the natural atmospheric gradients that determine rainfall volume but also its chemical composition. This is important, as specific chemicals dissolved in water can tell us information about its origin and even the altitude at which the precipitation that now flows through a stream fell. Soil geochemical and soil moisture fingerprints from next-generation radiometric and satellite datasets are used to guide the representation of the erodibility of the Northland landscape, but also where and how much organic carbon (peat) is stored below the ground surface. Such information is critical for understanding why the quality of water changes from stream to stream or from place to place along a stream.

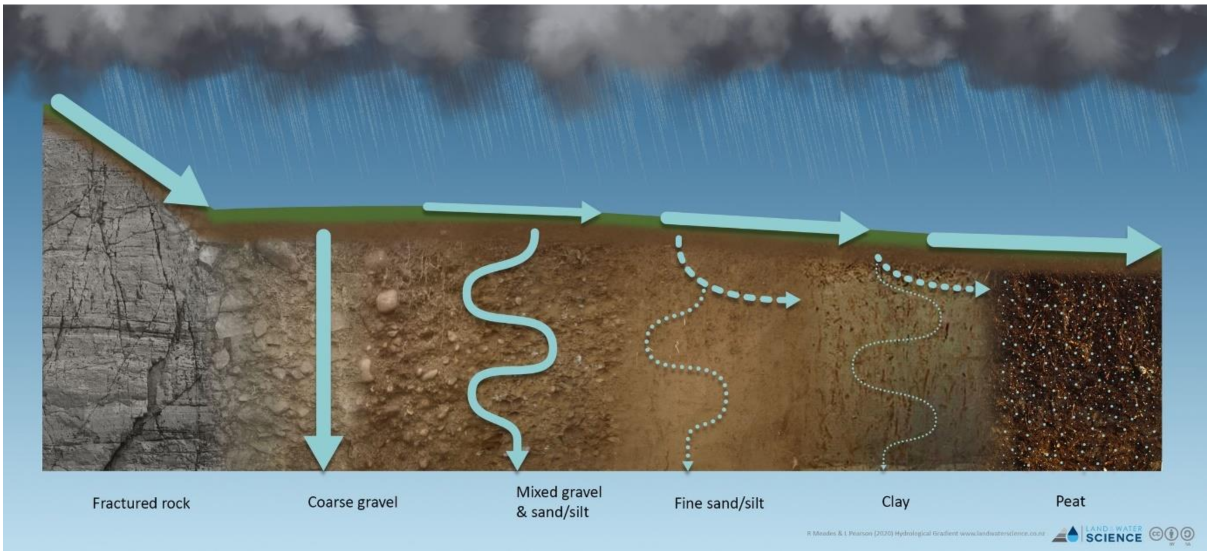


Figure 5. A simplified process-attribute gradient depicting the different hydrological pathway (response) water takes as slope, soil permeability, and drainage class vary.

Physiographic science maps the relationship between the processes controlling water quality (atmospheric, hydrological, redox, chemical and physical weathering) using information about landscape characteristics (e.g. soil, geology, topography) and the chemistry of water. We name the maps of each of the dominant processes that influence water quality process-attribute gradients

(PAG). Each of these maps attempts to replicate the natural gradients of the landscape, that in conjunction with land use, govern the spatial variation in water quality. We also map the gradients of land use intensity to account for the variation in land use across the region. The main foundation of the physiographic modelling is the 15 landscape PAG and 2 Land use PAG that were generated for the region (Rissmann et al., 2018a, Rissmann et al., 2018b; Rissmann et al., 2019a,b,c, McDonald et al., 2020) and used for physiographic modelling of Northland's steady-state water quality and load estimation.

3.1.1 Steady State Water Quality Modelling

This section provides a summary of the modelling undertaken in Rissmann and Pearson (2020) to predict water quality for unmonitored waterways in the Northland Region. Here water quality data from 67 water quality monitoring sites between January 2015 and December 2019 was used to model water quality across the river network (RECv1). The main objective of the statistical modelling approach is to generate a transparent mathematical model of the 'best' relationship between PAG and/or land use and chemical fingerprint or water quality measure. This enables the user to identify the PAG or PAG combination which explains the majority of the water quality variation across the region. For example, clarity decreases and turbidity increases as the erosion susceptibility risk increases. Rissmann and Pearson (2020) contains the sensitivities and magnitude of PAG responses for each model and are useful tools in identifying the factors controlling water quality.

There are two stages of modelling. The first uses the chemical fingerprints in water to test that the PAG do indeed represent the gradients in each dominant process. The second is dependent on the 1st stage and is when land use is incorporated to generate steady-state models of individual water quality measures. Both modelling stages employ the same machine learning or statistical modelling method that generates so-called "white box" models. These models differ from traditional "black box" models that are opaque and lack transparency about a model outcome – the model is not interpretable. The white box modelling employed is also smart enough to discard any PAG (or land use layer) that are not important predictors of a chemical fingerprint or water quality measure. It does this through billions of computations, that search for the best possible combination of PAG and/or land use that maximise accuracy and minimise the complexity of the resultant model. This type of modelling is also referred to as an 'evolutionary modelling approach' as it ultimately searches for the 'fittest' model for a given chemical fingerprint or water quality measure. During the evolution of a model, the PAG and/or land use layers that are the most sensitive predictors are retained. Any PAG that do not both improve the accuracy and reduce the complexity of the model is discarded. The PAG retained are often predictable according to the chemical or water quality contaminant being modelled. Overly complicated, black-box models are considered by some to be a poor substitute for understanding and representing the important role of the landscape over water composition and quality.

The River Environment Classification (REC) version 2.4 (Snelder and Biggs, 2002) was used to generate capture zones (watersheds) for each of the 67 long-term monitoring sites. The capture zones were then used to calculate mean scores for each of the 15 PAG and two land use PAG. These scores were then joined with median hydrochemical and water quality data to develop models of steady-state Total Nitrogen (TN), Ammoniacal Nitrogen (NH_4-N), Nitrate Nitrogen (NO_3-N), Dissolved Inorganic Nitrogen (DIN), Total Phosphorus (TP), Dissolved Reactive Phosphorus (DRP), microbial contamination as indicated by *E. coli,* Total Suspended Solids (TSS), clarity, and turbidity. The models developed use what is called 'cross-validation' to assess performance. Simply, the modelling approach withholds a large number of the sites, builds a model on the few sites and then tests to see if the model that has been built does a reasonable job of estimating the median values for all the withheld sites. The model switches the sites for model development in and out, after billions of calculations, seeking the best combination of sites, land use, and PAGs that best estimate an individual water quality measure. The 95th percentile and maximum values were also modelled for the Northland Region's State of Environment surface water monitoring network.

3.1.2 Load Estimation

Load refers to the total mass passing a river monitoring site over a given period (e.g., a month, year). Both concentration and flow are continuous functions of time; however, they are seldom measured continuously. Therefore, catchment contaminant loads are estimated from calculations that combine discrete contaminant concentration observations with more frequent observations of flow (Dolan et al., 1981; Cohn et al., 1989; Slender et al., 2014, 2017). Many approaches have been developed to calculate loads in the absence of continuous concentration measurements (Beale, 1962; Moatar and Meybeck, 2005; Quilbé et al., 2006; Snelder et al., 2014; Nava et al., 2019). Generally, there are three methods to estimate load: simple aggregation/interpolation techniques, ratio estimators, and regression-based techniques (Nava et al., 2019).

By summing the product of concentration and flow, we can quantify and compile the load at smaller time intervals. River flow can be measured by frequent stage measurements (e.g., hourly) from river monitoring sites. As water quality measurements can be expensive to apply, the data is usually collected less frequently. Without a continuous time-series of water quality data, the concentration of water quality compounds is estimated from flow (Lee et al., 2016).

The general load calculation equation is:

$$L = \int_{t_1}^{t_2} Q(t)C(t)\,dt \qquad \text{(Equation 1)}$$

Where L refers to the load between the period t_1 and t_2, Q(t) refers to the flow at time t, and C(t) refers to the concentration at time t. For the units, Q is in $m^3\ s^{-1}$ and C is in $g\ m^{-3}$. The time t is in seconds(s), and the result L is in grams (Quilbé et al., 2006).

To estimate load across the Northland region, we used a 5-year time period (2015-2019) and applied nine of the most common compound river load algorithms to each of Northland's 31 continuous flow and water quality sites using the R code of Nava et al. (2019). An additional four methods were also applied from Snelder et al. (2014, 2017). To model load across the River Environment Classification (REC1), the median load estimate was area weighted and modelled using the physiographic process attribute gradient classification (Rissmann and Pearson, 2020, Pearson et al., 2020).

3.2 Regional Modelling Results

3.2.1 Steady State Water Quality Model

The final models for each water quality measure are a series of mathematical equations that combined different PAG, land use, and mathematical functions. Table 2 presents the performance of the models for the 67 regional monitoring sites. The resulting equations (models) can then be used to estimate water quality across unmonitored stream reaches. The modelled concentrations for median and Q95 for TN, TP, TSS, and *E.coli* are shown across the Hokianga Harbour river network (Figures 6 & 7).

Given the calibrations and development of the models at stream orders greater than 2, it is expected that there will be some limitations for lower stream orders. An assessment of the measured and modelled concentrations was very good to reasonable for most water quality species (see section 4).

Table 2. Model performance measures for Northland's river water quality (5-year median concentration).

	Nitrogen					Phosphorus		Sediment			Microbial
	TN	NO_3-N	DIN	TKN	TAM	TP	DRP	TSS	Turb.	Clarity	*E. coli*
Cross-validated R^2	0.86	0.80	0.84	0.75	0.76	0.67	0.69	0.70	0.79	0.65	0.61
Correlation Coefficient	0.93	0.90	0.92	0.87	0.87	0.82	0.83	0.84	0.89	0.81	0.79
Maximum Error	0.38	0.85	0.77	0.42	0.35	0.52	0.39	0.40	0.38	0.25	0.46
Mean Squared Error	0.02	0.08	0.05	0.01	0.01	0.02	0.02	0.01	0.01	0.01	0.02
Mean Absolute Error	0.10	0.20	0.17	0.06	0.08	0.09	0.10	0.08	0.07	0.05	0.11
Coefficients	8	5	8	8	7	6	6	7	6	7	7
Complexity	35	46	51	38	36	41	58	98	38	98	33

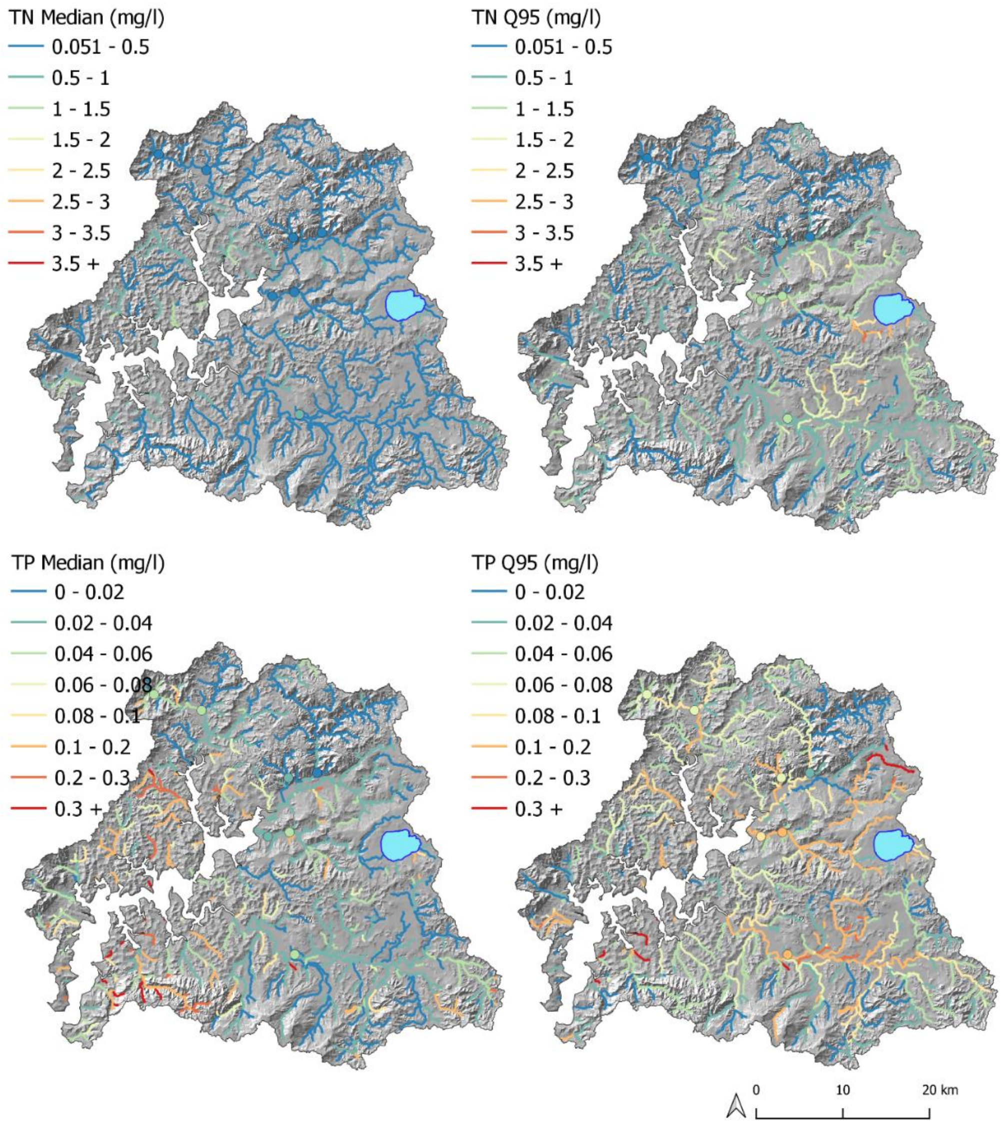


Figure 6. Modelled median Total Nitrogen (TN) (top left), 95th percentile TN (top right), median Total Phosphorus (TP) (bottom left) and 95th percentile TP (bottom right) in Hokianga rivers. Circles denote the observed median and Q95 values for each of the six monitoring sites within the catchment. River reaches and observed measures (sites) are colour coded according to the same concentration gradient in mg/l.

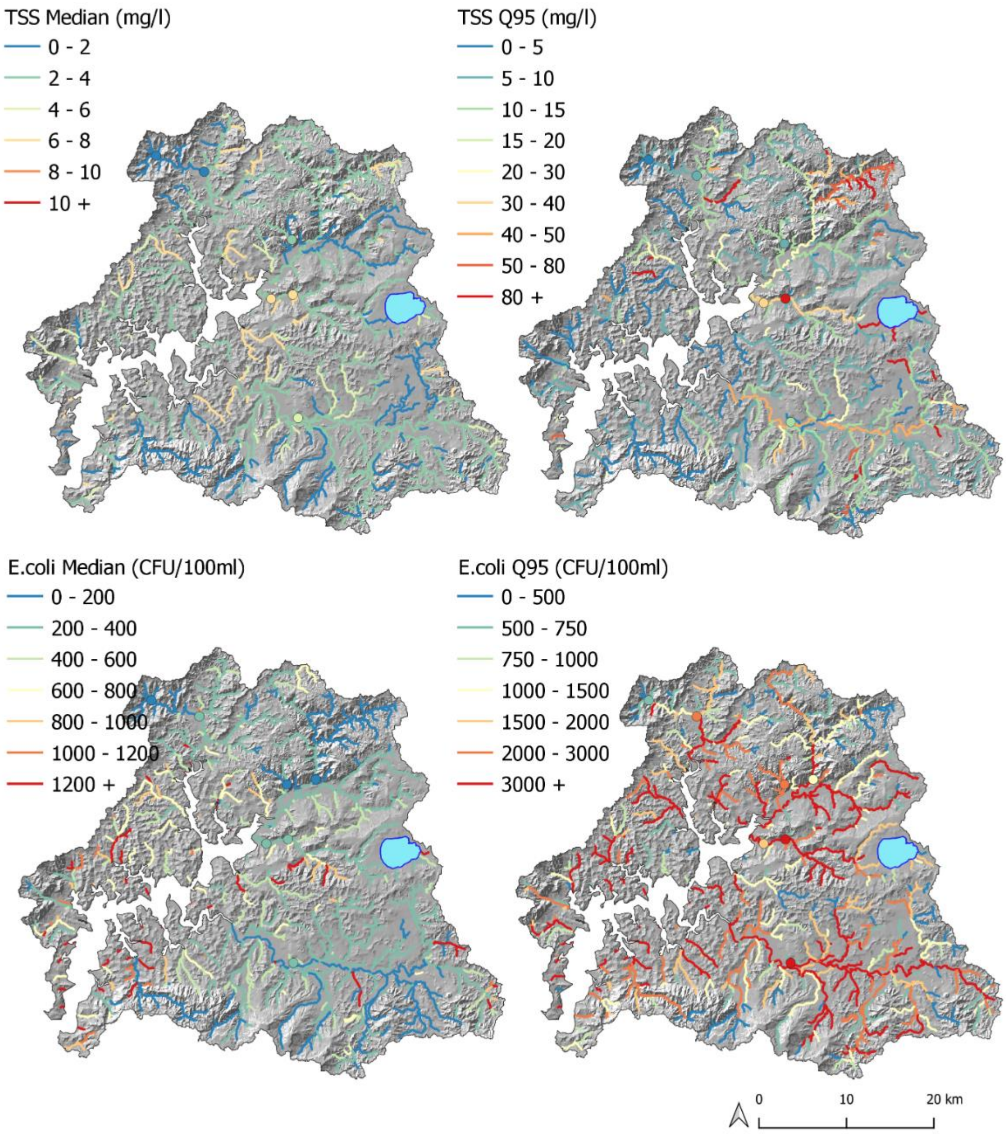


Figure 7. Predicted median Total Suspended Sediment (TSS) (top left), 95th percentile TSS (top right), median E.coli (bottom left) and 95th percentile E.coli (bottom right) in Hokianga rivers. Circles denote the observed median and Q95 values for each of the monitoring sites. River reaches and observed measures (sites) are colour coded according to the same concentration gradient in mg/l and CFU/100ml for E.coli.

The models of steady-state water quality across the digital stream network are consistent with the underlying physiographic mapping of the Northland Region (Rissmann et al.,2018a,b; 2019b,c). Specifically, they indicate a robust spatial linkage between landscape attributes and hydrochemical signatures of the dominant processes and associated water quality outcomes. Some of the key findings of this work include:

- Nitrate nitrogen (NO_3-N), constitutes a relatively small fraction of the total TN concentration across Northland's monitoring sites (1/3rd of the TN concentration on average, Figure 6). Overall, NO_3-N levels are low by national standards, with reduced nitrogen species, i.e., ammoniacal and organic nitrogen, constituting the bulk of the load exported to streams. The loading of organic and ammoniacal nitrogen to streams, lakes, and harbours contributes to the store of potential mineralisable nitrogen in benthic sediments. Here it is important to note that organic and ammoniacal forms of nitrogen ultimately end up being mineralised to nitrate and nitrite.

- Dissolved reactive phosphorus (DRP) exhibits a strong geological control with the highest concentrations associated with steep outcrops of Tangihua Volcanics (Figure 6). Basalt commonly contains higher elemental phosphorus concentrations than felsic sedimentary rock and weathers faster than felsic rock, supplying inorganic P to the river network. The spatial correlation between elevated DRP and Tangihua Volcanic Complex extends from Cape Reinga in the North to Tangihua Forest in the south, wherever the unit outcrops. However, it is also notable that there is a positive correlation between terrain ruggedness and DRP concentration derived from the Tangihua Volcanic Complex. Notably, the flat-lying Waipoua and Kerikeri flood basalts appear less implicated in DRP generation. Perhaps due to lower terrain ruggedness, mantling by siliceous materials and the development of a stable soil mantle. Other areas of moderately elevated DRP occur in association with peat and lacustrine sediments.
- Total phosphorus (TP) exhibits similar geological associations to DRP with respects to the Tangihua Volcanics and the peat and lacustrine rich portions of the Tauranga Formation and Karioitahi Group (Figure 6). There is also evidence that land use and poorly drained soils play an important role in the distribution of the particulate phosphorus (PP) fraction of TP. This is consistent with overland flow and artificial drainage density being retained by the model, in addition to geological PAG. Further, PP is known to show a strong association with developed land with dissolved organic and inorganic forms more commonly associated with natural state settings.
- There is a strong geological correlation between elevated sediment (i.e., turbidity, clarity and total suspended sediment) and soft and highly erodible lithologies as defined by the Erosion Susceptibility Classification (ESC) of Rissmann et al. (2018b) (Figure 7). For example, the poorly lithified weak sedimentary rocks of the Northland Allochthon, including but not limited to the Punakitere Sandstone. Sediment is also elevated in relationship to depositional landforms, i.e., alluvium, peat, and lacustrine sediments of the Tauranga Group and Karioitahi Group, especially where the water table is shallow and soils poorly drained. Streams draining harder lithologies, such as the rocks of the Tangihua Volcanic Complex and Waipoua Basalt, show lower turbidity. Also notable, is that estimated sediment concentration is low across the areas of well-drained soils where surficial runoff and artificial drainage is less prevalent (e.g. across a significant area of the low relief Kerikeri flood basalts). Tributaries of the Wairoa River, including the Manganui, are identified as being particularly sediment rich.
- Highest median *E. coli* counts coincide with areas of erosion-prone land that has been developed for extensive or intensive land use (Figure 7). *E. coli* is also elevated across depositional landforms (floodplains etc.) where soils are poorly drained, and the local water table is shallow. For example, the majority of elevated *E. coli* counts coincide with developed sheep, beef and dairying land on highly erodible land not limited to the Punakitere Sandstone and other soft sedimentary rocks of the Northland Allochthon. The Karioitahi Group and Tauranga Group sediments are also implicated, especially where the water table is elevated, and soils are poorly drained.
- Land use was also implicated in water quality, but overall, the influence of landscape factors was more important than land use on its own. However, due to the correlation between land use and landscape attributes, it is likely that some of the PAG are acting as surrogates for land use intensity (e.g. artificial drainage PAG). Issues of correlation can be addressed through the refinement of the land use layer and additional statistical treatment.

3.2.2 Regional Load Estimates

Thirteen load algorithms were calculated for each site and each main contaminant using flow and water quality data for a 5-year period between January 2015 and December 2019. Estimates of load were generated for Total Nitrogen (TN), Nitrate Nitrogen (NO_3-N), Nitrate-Nitrite Nitrogen (NNN), Total Ammoniacal Nitrogen (TAM), Total Kjeldahl Nitrogen (TKN), Total Phosphorus (TP), Dissolved Reactive Phosphorus (DRP), Total Suspended Solids (TSS), and microbial contamination as indicated by *E.* coli for each of the 31 sites. There was good agreement in the magnitude of load estimates between the algorithms of Nava et al. (2019) and Snelder et al. (2014, 2017). Median loads from all 13 methods were normalised by capture zone area to produce export coefficients for nutrients and sediment in kg/ha/yr, and *E.coli* in CFU 10^9/ha/yr for each contaminant at all 31 monitoring sites.

Models of the relationship between export coefficients, physiographic layers (process-attribute gradients) and land use were then developed to estimate load across the wider Northland region. Export coefficient modelling followed the same method for steady-state modelling (Rissmann and Pearson, 2020, Pearson et al., 2020), employing a hybrid deterministic genetic programming (HDGP) approach of Schmidt and Lipson, (2009) that is specifically designed to identify the underlying drivers of variation in complex non-linear systems (Bongard and Lipson, 2005, 2007; Schmidt and Lipson, 2008, 2010, 2011, 2015). Performance statistics of the regional load models are provided in Table 3 (Pearson et al., 2020).

Overall the modelled load performance was good with cross-validated R^2 values between 0.86 and 0.88 for nitrogen species, 0.77 to 0.92 for phosphorus species, 0.78 for sediment, and 0.72 for *E.coli* (Table 3). It is expected that models of dissolved species (TN, NO_3-N, NNN, TKN, TAM, DRP) perform better than those represented by particulate transport (TSS, *E.coli*) or both (TP) as the processes controlling contaminant loss are relatively easier to predict. Although the DRP model has a high correlation coefficient, the complexity of the model is significantly higher than the other models, which can suggest multiple variables controlling DRP concentration in the region.

Table 3. Regional load model performance.

	Nitrogen					Phosphorus		Sediment	Microbial
	TN	NO_3-N	NNN	TKN	TAM	TP	DRP	TSS	*E. coli*
R^2	0.88	0.87	0.88	0.86	0.86	0.77	0.92	0.78	0.72
Correlation Coefficient	0.94	0.93	0.94	0.93	0.93	0.88	0.96	0.88	0.86
Maximum Error	0.28	0.48	0.46	0.22	0.20	0.32	0.27	0.37	0.61
Mean Squared Error	0.01	0.03	0.02	0.01	0.01	0.01	0.01	0.01	0.04
Mean Absolute Error	0.06	0.11	0.11	0.05	0.05	0.07	0.05	0.08	0.14
Coefficients	6	6	7	6	7	8	6	7	6
Complexity	42	27	32	31	35	49	114	40	43

The patterns of contaminant export are broadly consistent with those observed in the steady-state modelling and indicate that both landscape factors and land use govern contaminant export across the region. Overall, the physiographic layers retained as the most sensitive predictors of contaminant export were similar to those retained by steady-state models. More confidence is given to the steady-state depiction of landscape controls, given the larger number of sites used for model development (n = 67).

4 Hokianga Catchment Water Quality State

Measured water quality data for a 5-year period (2015-2019) was used to assess water quality state against the National Policy Statement for Freshwater Management National Objectives Framework (Ministry for the Environment 2020). For comparison with the measured data, the modelled value and modelled state are provided for all attributes. Attributes are assessed for either ecosystem health (water quality) or human contact (i.e. recreation) purposes. The NOF state of the six sites within the Hokianga Harbour catchment is summarised below.

This assessment was undertaken while the Ministry of Environment was in the process of updating at the National Objectives Framework (NOF). A draft National Policy Statement for Freshwater Management, released in September 2019, was proposed as a full replacement of the National Policy Statement for Freshwater Management 2014 (as amended 2017). This consultation draft introduced new national objectives for ecosystem health for dissolved inorganic nitrogen (DIN) and dissolved reactive phosphorus (DRP). Following consultation, the National Policy Statement for Freshwater Management 2020 was released in August. This final version removed the requirement for DIN to be assessed by regional authorities and introduced a new attribute of visual clarity to assess suspended fine sediment to the NOF. Therefore, this assessment may differ slightly from that previously reported in Rissmann and Pearson (2020), which was completed prior to the release of the NPSFM 2020.

4.1 Ecosystem Health

The NOF in the NPSFM (2019, 2020) contains the following indicators of ecosystem health. These include attributes for both indicators for toxic effects (nitrate and ammonium toxicity) and trophic state (dissolved inorganic nitrogen, dissolved reactive phosphorus, and suspended fine sediment). The state of the water quality monitoring sites within the Hokianga Harbour is as follows.

4.1.1 Nitrate Toxicity

Table 4 shows all sites within the Hokianga FMU are within the A band (≤1.0 mg/l annual median and ≤1.5mg/l annual 95th percentile) of the NOF for nitrate toxicity. This attribute measures the toxic effects of nitrate (not trophic state). At this state, there is unlikely to be effects even on sensitive species (NPSFM, 2020). Attribute state was correctly modelled for all sites within the Hokianga catchment.

Table 4. Assessment against the National Objectives Framework for nitrate toxicity (NO_3-N mg/l) for measured and modelled data (2015-2019).

Site Name	Measured median conc.	Measured median state	Model median conc.	Model median state	Measured 95th percentile	Measured 95th percentile state	Model 95th percentile	Model 95th percentile state
Mangamuka at Iwitaua Road	0.005	A	0.016	A	0.061	A	0.054	A
Tapapa at SH1	0.026	A	0.021	A	0.066	A	0.047	A
Waipapa at Forest Ranger	0.030	A	0.015	A	0.066	A	0.067	A
Utakura at Horeke Rd	0.110	A	0.190	A	0.242	A	0.690	A
Utakura at Okaka Bridge	0.155	A	0.152	A	0.313	A	0.700	A
Punakitere at Taheke	0.375	A	0.137	A	0.583	A	0.553	A

4.1.2 Ammonia Toxicity

Ammonia toxicity state is shown in Table 5 for both measured and modelled data. Annual median values have an attribute state of A band (≤0.03 mg/l) which means there is no observed effect on any species tested. For the annual TAM maximum state, the predominantly natural state sites were in the A band (≤0.05mg/l), while Punakaitere at Taheke and Utakura at Horeke Rd are in the B band (>0.05mg/l and ≤0.40mg/l). Utakura at Okaka Bridge, which has a land use predominantly in pastoral farming, is in the C band (>0.40 mg/l and ≤2.20 mg/l). In the B attribute state, there are occasional impacts on the 5% most sensitive species, which increases to regular impacts on 20% of the most sensitive species in state C (NPSFM, 2020). Attribute state was correctly modelled for all median values, while the annual maximum was overestimated at Punakitere at Taheke and Utakura at Horeke Rd sites.

Table 5. Assessment against the National Objectives Framework for ammonia toxicity (TAM mg/l) measured and modelled data (2015-2019).

Site Name	Measured median conc.	Measured median state	Model median conc.	Model median state	Measured annual maximum	Measured annual maximum state	Model annual maximum	Model annual maximum state
Waipapa at Forest Ranger	0.004	A	0.005	A	0.008	A	0.026	A
Tapapa at SH1	0.005	A	0.005	A	0.018	A	0.021	A
Mangamuka at Iwitaua Road	0.005	A	0.006	A	0.037	A	0.027	A
Punakitere at Taheke	0.012	A	0.011	A	0.180	B	0.427	C
Utakura at Horeke Rd	0.014	A	0.014	A	0.059	B	0.454	C
Utakura at Okaka Bridge	0.015	A	0.016	A	0.430	C	0.461	C

4.1.3 Dissolved Inorganic Nitrogen

Although DIN is no longer an attribute in the NPSFM 2020, it was included in this assessment for comparison with DRP as enrichment in both nutrients are likely to favour eutrophication. DIN provides an indication of ecosystem trophic state. In the Hokianga catchment, most sites are band A, and Punakitere at Taheke is band B (Table 6). In band A (median ≤0.24 mg/l and 95th percentile ≤0.56 mg/l) the ecological communities and ecosystem processes are similar to those of natural reference conditions. No adverse effects attributable to DIN enrichment are expected at this state. For Punakitere at Taheke in band B, there may be some slight impact to ecological communities from minor DIN elevation above natural reference conditions. If conditions also favour eutrophication, sensitive ecosystems may experience additional algal and plant growth, loss of sensitive macroinvertebrate taxa, and higher respiration and decay rates (NPSFM, 2019).

Table 6. Assessment against the proposed National Objectives Framework (2019) for dissolved inorganic nitrogen measured and modelled data (2015-2019). DIN was calculated from the sum of NO_3-N and NO_2-N.

	Measured median conc.	Measured median state	Model median conc.	Model median state	Measured 95th percentile	Measured 95th percentile state	Model 95th percentile	Model 95th percentile state
Mangamuka at Iwitaua Road	0.012	A	0.011	A	0.067	A	0.067	A
Waipapa at Forest Ranger	0.015	A	0.002	A	0.086	A	0.088	A
Tapapa at SH1	0.031	A	0.038	A	0.072	A	0.089	A
Utakura at Horeke Rd	0.115	A	0.224	A	0.274	A	0.300	A
Utakura at Okaka Bridge	0.167	A	0.318	B	0.339	A	0.310	A
Punakitere at Taheke	0.398	B	0.176	A	0.616	B	0.616	B

4.1.4 Dissolved Reactive Phosphorus

In the Hokianga catchment, the sites have a range of DRP state from A at Waipapa at Forest Ranger to D at Tapapa at SH1 (Table 7). Although DRP may be elevated, on its own it isn't sufficient to favour eutrophication and requires DIN to also be elevated to increase algal and plant growth and the other effects noted above. The newly released NPSFM (2020) provides exemptions for naturally occurring processes under section 3.32, with the regional council required to demonstrate that it is naturally occurring processes that prevents the national bottom line being achieved. The ability to predict attribute state for DRP is significantly more challenging due to the narrow concentration bands associated with this attribute, and overall modelled concentrations are generally pretty accurate for median concentrations and less so for 95th percentile estimates.

Sites can be classified in the C or D band despite being predominantly natural state. As identified in the regional steady state modelling, there is a strong geological control over DRP with the highest concentrations associated with steep outcrops of Tangihua Volcanics. Petrological assays of Tangihua Volcanics report inorganic P concentrations that range from 1,000 – 6,000 ppm with a median of 2,500 ppm based on 20 measures (Briggs and Searle, 1975; Nicholson et al., 2000). These concentrations sit at the higher end of P-bearing rocks globally with a range of 120 - >3,200 ppm reported by Porder and Ramachandran (2013) for over 2,500 samples of common rock types globally. These authors reported median P concentrations for common silica-rich rocks of 500 ppm and 1,000 ppm for common iron-rich rocks (e.g. andesite - 1,000 ppm).

Table 7. Assessment against the proposed National Objectives Framework for dissolved reactive phosphorus (DRP mg/l) for measured and modelled data (2015-2019).

	Measured median conc.	Measured median state	Model median conc.	Model median state	Measured 95th percentile	Measured 95th percentile state	Model 95th percentile	Model 95th percentile state
Waipapa at Forest Ranger	0.005	A	0.004	A	0.007	A	0.015	A
Utakura at Horeke Rd	0.010	B	0.011	C	0.025	B	0.020	A
Utakura at Okaka Bridge	0.013	C	0.011	C	0.031	C	0.020	A
Punakitere at Taheke	0.023	D	0.010	B	0.057	D	0.018	A
Mangamuka at Iwitaua Road	0.034	D	0.019	D	0.042	C	0.041	C
Tapapa at SH1	0.036	D	0.038	D	0.060	D	0.053	C

4.1.5 Suspended Fine Sediment

Suspended fine sediment was added as an attribute for assessing ecosystem health in the NPSFM 2020. Climate, water source, and geology were identified at the monitoring site from the REC to identify the class to assess the numeric attribute state against (Table 8). Tapapa at SH1 and Waipapa at Forest Ranger are in band A which means there is minimal impact from suspended sediment on instream biota. Mangamuka at Iwitua Road and Punakitere at Taheke are classified as band B which means there is low to moderate impact of suspended sediment on instream biota. The abundance of sensitive fish species may be reduced. Sites Utakura at Okaka Bridge and Utakura at Horeke Road are lake fed and although they have similar clarity to other sites in the catchment, the assessment criteria for lake fed rivers is much higher, with the national bottom line set at > 2.22 meters. In the D band there is expected to be a high impact of suspended sediment on instream biota. Ecological communities are significantly altered, and sensitive fish and macroinvertebrate species are lost or at high risk of being lost. However, there may be naturally occurring processes that are influencing the state of these sites as autochthonous phytoplankton production can produce low visual clarity in warm climates, such as Northland. The modelled concentrations were generally pretty accurate for predicting median clarity and therefore attribute state for suspended fine sediment (Table 8).

Table 8. Assessment against the National Objectives Framework for suspended fine sediment (clarity in meters) for ecosystem health (2015-2019). Sites with an overall state indicated with an asterisk (*) do not meet the minimum sample number requirements and are to be taken as indicative only.

Site Name	REC	Class	No. of samples	Median Clarity (m)	Measured Median State	Model Clarity (m)	Model Median State
Tapapa at SH1	WW-Low-VA	1	42*	1.90	A	1.94	A
Mangamuka at Iwitaua Road	WW-Low-VA	1	60	1.50	B	1.75	B
Waipapa at Forest Ranger	WW-Low-SS	2	57*	3.23	A	1.14	A
Punakitere at Taheke	WW-Low-SS	2	55*	0.91	B	1.29	A
Utakura at Okaka Bridge	WW-Lake-SS	3	40*	1.10	D	1.11	D
Utakura at Horeke Rd	WW-Lake-HS	3	69	0.94	D	1.09	D

Note: REC classification WW – warm wet climate, Low – lowland water source, Lake – Lake fed water source, VA – volcanic geology, SS – Soft sedimentary geology, HS- Hard sedimentary geology.

4.2 Human Health for Recreation

E.coli is used as an indicator to assess the risk to human health from Campylobacter infection. Attribute state is determined by satisfying all numeric attribute states according to median concentration, 95th percentile, percentage exceedances over 540 *E.coli*/100ml and percentage exceedances over 260 E.coli/100ml (Table 9). At Tapapa at SH1 for at least half the time, the estimated risk is <1 in 1000 (0.1% risk) with a predicted infection rate of 2% (B band). At Waipapa at Forest Ranger, the risk is 0.1% with a predicted average infection risk of 3% (C band). Both sites on the Utakura River are in the D band, 20-30% of the time the estimated risk is ≥50 in 1000 (>5% risk) with a predicted average infection risk of >3%. Managamuku at Iwitaua Road and Punakitere at Taheke are both in the E band where for more than 30% of the time the estimated risk is ≥50 in 1000 (>5% risk) with a predicted average infection risk of > 7%. Predicted average infection risk is the overall average infection to swimmers based on a random day. Actual risk will generally be less if a person does not swim during high flows. The modelled state is provided in Table 10. However, due to the inability to model percentage exceedances greater than 260 and 540 CFU/100ml, an overall state is not able to be modelled.

Table 9. Assessment against the National Objectives Framework for E.coli (measured) Human Health for Recreation.

Site Name	Number of samples	Median (cfu/100 ml)	Median State	95th percentile (cfu/100 ml)	Q95 State
Tapapa at SH1	42	103.5	A	581.4	B
Waipapa at Forest Ranger	60	81.1	A	1124.1	C
Utakura at Horeke Rd	72	206.0	D	1509.9	D
Utakura at Okaka Bridge	47	250.0	D	4953.3	D
Mangamuka at Iwitaua Road	60	345.0	E	2938.6	D
Punakitere at Taheke	60	327.0	E	5863.4	D

Table 9 continued. Assessment against the National Objectives Framework for E.coli (measured) Human Health for Recreation. Sites with an overall state indicated with an asterisk (*) do not meet the minimum sample number requirements and are to be taken as indicative only.

Site Name	*E. coli* >260 cfu/ 100 ml	% >260 cfu/ 100 ml	% >260 State	*E. coli* >540 cfu/ 100 ml	% >540 cfu/ 100 ml	% >540 State	Overall State
Tapapa at SH1	7	16.7	A	3	7.1	B	B*
Waipapa at Forest Ranger	11	18.3	A	6	10.0	B	C
Utakura at Horeke Rd	28	38.9	D	8	11.1	C	D
Utakura at Okaka Bridge	21	44.7	D	12	25.5	D	D*
Mangamuka at Iwitaua Road	40	66.7	E	23	38.3	E	E
Punakitere at Taheke	36	60.0	E	24	40.0	E	E

Table 10. Assessment against the National Objectives Framework for E. coli modelled data. Percent exceedances greater than 260 and 540 CFU/100ml are unable to be modelled.

Site Name	Model Median (cfu/ 100 ml)	Model Median State	Model Q95 (cfu/ 100 ml)	Model State
Tapapa at SH1	95	A	379	A
Waipapa at Forest Ranger	115	A	1485	D
Utakura at Horeke Rd	272	E	4136	D
Utakura at Okaka Bridge	262	E	3915	D
Mangamuka at Iwitaua Road	166	D	1772	D
Punakitere at Taheke	166	D	3497	D

5 Hokianga Harbour Loads Estimate

5.1 Load Estimate at Water Quality Monitoring Sites

Of the monitoring sites within the Hokianga harbour catchment, Punakitere at Taheke and Waipapa at Forest Ranger had associated flow data to estimate contaminant load. Load was calculated by applying nine methods from the RiverLoad script of Nava et al. (2019) and four methods from Fraser and Snelder (2020) using flow and measured water quality data collected over a 5-year period between January 2015 and December 2019.

The selection of the 'best' method for load estimation is often highly subjective (Fraser and Snelder, 2020), a median load ('integrated load') was calculated from all 13 load estimates for each site and each contaminant. As a means of quality control, Hierarchical Cluster Analysis (HCA) was applied to each contaminant per site to group the estimated loads according to magnitude. HCA revealed which of the 13 different algorithms provide similar estimates and identified outliers. As there were few outliers, a median or 'integrated' load estimate was determined using all 13 different load estimates by site and contaminant. Grouping the load estimates this way provides an integrated measure of likely load that has the benefit of a confidence interval and associated range for each site. See Pearson et al. (2020) for a detailed description of the methods applied. Outliers were removed from the calculation of mean scores and associated descriptive statistics and confidence intervals (95%). Load estimates from all methods and summary statistics for Punakitere at Taheke and Waipapa at Forest Ranger are shown in Tables 11 and 12, and Tables 13 and 14, respectively.

Table 11. Load estimate for Punakitere at Taheke in tonnes per year and CFU 10^9 per year for E.coli from all methods. The cluster membership of the load equations is indicated by shading for each method (yellow – largest cluster, blue – 2nd cluster and orange – 3rd cluster). The methods of Fraser and Snelder (2020) were unable to be applied to E.coli loads.

Analyte	Method 1	Method 2	Method 3	Method 4	Method 5	Method 6	Beale ratio	Rating	Ferguson	L7	L5	FS	FST
TN	381.96	459.63	501.9	718.55	864.66	859.95	876.88	1228.8	1326.12	1084.98	1115.85	769.57	657.28
NNN	186.98	207.16	428.19	351.74	389.72	428.95	388.37	1078.7	1924.87	894.71	1803.37	371.89	230.74
TAM	13.116	19.316	29.048	24.673	36.338	24.111	37.499	67.981	85.796	43.518	35.811	20.516	14.065
TKN	198.65	255.77	536.82	373.71	481.15	431.95	494.7	573.73	597.48	774.28	586.54	398.12	325.46
TP	27.57	41.93	75.02	51.87	78.89	86.33	82.17	120.52	136.58	169.78	135.04	75.19	50.47
DRP	14.65	22.16	13.39	27.56	41.68	30.76	43.35	65.61	74.35	36.21	40.12	28.55	23.55
TSS	2239.63	4501.62	4109.62	4213.2	8468.49	2679.61	9017.29	15784.82	18984.76	64626.3	69457.4	752.57	2390.4
E.coli	3041514	18846854	3200832	2803764	17373626	3628336	19258208	1956775.2	3626296				

Table 12. Load statistics (median, mean, minimum, maximum, range, 95% confidence interval, standard deviation, and coefficient of variation) at Punakitere at Taheke. Median is calculated from all available load methods. Method identifies the load calculation method, which corresponds to the median value. No of methods is the count of methods used to calculate the summary statistics (except for median values).

Analyte	Median	Method	Mean	Minimum	Maximum	Range	95% C.I.	S.D.	C.V.	No. Methods	Mean Cluster 1	Mean Cluster 2	Mean Cluster 3
TN	859.95	Method 6	834.32	381.96	1326.12	944.16	154.3	283.85	34.02	13	791.15	1188.94	447.83
NNN	389.72	Method 5	668.11	186.98	1924.87	1737.89	308.31	567.17	84.89	13	331.53	1425.41	
TAM	29.048	Method 3	34.753	13.116	85.796	72.681	11.061	20.347	58.548	13	20.69	38.29	76.89
TKN	481.15	Method 5	463.72	198.65	774.28	575.62	81.77	150.43	32.44	13	577.81	330.61	
TP	78.89	Method 5	87.03	27.57	169.78	142.21	21.98	40.43	46.45	13	79.52	42.96	140.48
DRP	30.76	Method 6	35.53	13.39	74.35	60.97	9.44	17.36	48.85	13	24.96	36.81	69.98
TSS	4501.62	Method 2	15940.44	752.57	69457.4	68704.83	12186.57	22418.42	140.64	13	6649.27	67041.85	
E.coli	3626296	Ferguson	8192912	1956775	19258208	17301433	4777701	7312943	89.26	9	3042920	18492896	

Table 13. Load estimate for Waipapa at Forest Ranger in tonnes per year and CFU 10^9 per year for E.coli from all methods. The cluster membership of the load equations is indicated by shading for each method (yellow – largest cluster, blue – 2nd cluster and orange – 3rd cluster). Outliers identified by the clustering are indicated in green. The methods of Fraser and Snelder (2020) were unable to be applied to E.coli loads. TKN and TSS concentration data is unavailable for this site.

Analyte	Method 1	Method 2	Method 3	Method 4	Method 5	Method 6	Beale ratio	Rating	Ferguson	L7	L5	FS	FST
TN	69.22	78.15	63.68	76.06	85.87	91.55	86.65	166.55	176.7	402.35	153.97	33.85	65.27
NNN	16.3	20.65	29.04	17.91	22.68	19.04	23.2	457.88	525.73	34.9	146.37	6.48	8.54
TAM	2.555	2.668	3.083	2.807	2.931	2.643	2.944	3.225	3.425	3.288	2.76	1.022	2.175
TP	6.27	6.78	9.29	6.88	7.45	10.3	7.49	11.01	11.26	46.12	20.17	3.99	7.03
DRP	2.79	2.98	2.11	3.07	3.28	3.18	3.3	4.25	4.3	3.01	4.08	1.35	2.98
E.coli	330500	1135469	321913	354020	1216275	440431	1364743	646257	1071623				

Table 14. Load statistics (median, mean, minimum, maximum, range, 95% confidence interval, standard deviation, and coefficient of variation) at Waipapa at Forest Ranger. Median is calculated from all available load methods. The method identifies the load calculation method, which corresponds to the median value. No of methods is the count of methods used to calculate the summary statistics (except for median values).

Site	Median	Method	Mean	Minimum	Maximum	Range	95% C.I.	S.D.	C.V.	No. Methods	Mean Cluster 1	Mean Cluster 2	Mean Cluster 3
TN	85.87	Method 5	95.63	33.85	176.7	142.85	24.41	43.15	45.12	12	37.1	74.94	
NNN	22.68	Method 5	102.21	6.48	525.73	519.25	92.48	170.13	166.45	13	31.37	491.81	
TAM	2.807	Method 4	2.875	1.022	3.425	2.403	0.19	0.336	11.682	12	3.15	2.6	
TP	7.49	Beale.ratio	8.99	3.99	20.17	16.18	2.23	3.94	43.8	12	6.56	12.41	
DRP	3.07	Method 4	3.13	1.35	4.3	2.95	0.43	0.78	25	13	3.07	4.27	1.73
E.coli	646257	Rating	764581	321913	1364743	1042830	263863	403880	52.82	9	418624	1197027	

5.2 Regional Load Model applied to the Hokianga Harbour Catchment

To estimate load across areas of the digital network the load estimates were converted to an export coefficient or yield by dividing by the area. The export coefficients for the sites within the Hokianga are provided in Table 15. Punakitere at Taheke is predominantly developed for agricultural use while Waipapa at Forest Ranger is predominantly natural state.

Table 15. Export coefficient (kg ha^{-1} yr^{-1}) for sites within the Hokianga Harbour catchment.

Site Name	TN	TAM	TKN	NNN	TP	DRP	TSS	*E.coli* (CFU 10^9/ha/year)	Area (ha)
Punakitere at Taheke	26.36	0.89	14.75	11.95	2.42	0.94	138.01	111.17	32618.12
Waipapa at Forest Ranger	7.13	0.23	-	1.88	0.62	0.25	-	53.65	12046.90

Models of the relationship between export coefficients (yield), physiographic layers (process-attribute gradients) and land use were developed for the Northland region (Pearson et al., 2020). Export coefficient modelling followed the same method for steady-state modelling, employing a hybrid deterministic genetic programming (HDGP) approach of Schmidt and Lipson, (2009) that is specifically designed to identify the underlying drivers of variation in complex non-linear systems (Bongard and Lipson, 2005, 2007; Schmidt and Lipson, 2008, 2010, 2011, 2015).

Load estimates for the Hokianga have been calculated in two ways. The first uses the stream order capture zones which are located at the node points where two stream orders of the same magnitude converge. This allows the information to be layered by stream order to produce an output that can be interrogated to identify areas that contribute a high contaminant load (Figures 8-10). It is expected that there will be some limitations at low order reaches due to the small area relative to the calibration flow site capture areas. However, this approach provides more resolution over the sources to the harbour than assessing load on a subcatchment or catchment area. The second approach applied the model to the subcatchment areas to obtain a load estimate in kg yr^{-1} (Figure 11). As there are fewer small capture areas, the resulting output is likely to be more accurate but does not show the spatial resolution achieved above. The export coefficients were multiplied by the capture zone area (Ha) to calculate total load.

The total nitrogen (TN) load estimate appears to be mainly related to land use intensity and redox controls (Figure 8). Specifically, nitrate is lower where soils and aquifer materials are reducing as is evident across areas of peat wetland and in association with poorly drained soils along main floodplains. However, nitrate is a relatively minor contaminant in the Hokianga catchment. Like nitrate, total ammoniacal nitrogen (TAM) is also a minor component of the TN load. Regionally TAM exhibits a strong geological and soil-related control, with elevated concentrations associated with reducing bedrock and peat or organic soils. These settings are associated with high organic carbon content and reducing waters. Land use is also implicated, although it is overall a less sensitive predictor. Total Kjeldahl Nitrogen exhibits a strong geological and soil-related control with elevated concentrations associated with natural state forest, wetlands and organic soils. This is consistent with an organic nitrogen export. The importance of organic export over the catchments TN load is consistent with Regional SOE surface water quality monitoring data that indicates that at least 2/3^{rds} of the TN occurs in organic form, with nitrate making a minor proportion. The minor concentrations of nitrate in the Hokianga harbour catchments surface water is consistent with the large area of reducing soils, aquifers, and high organic carbon cover of forested land.

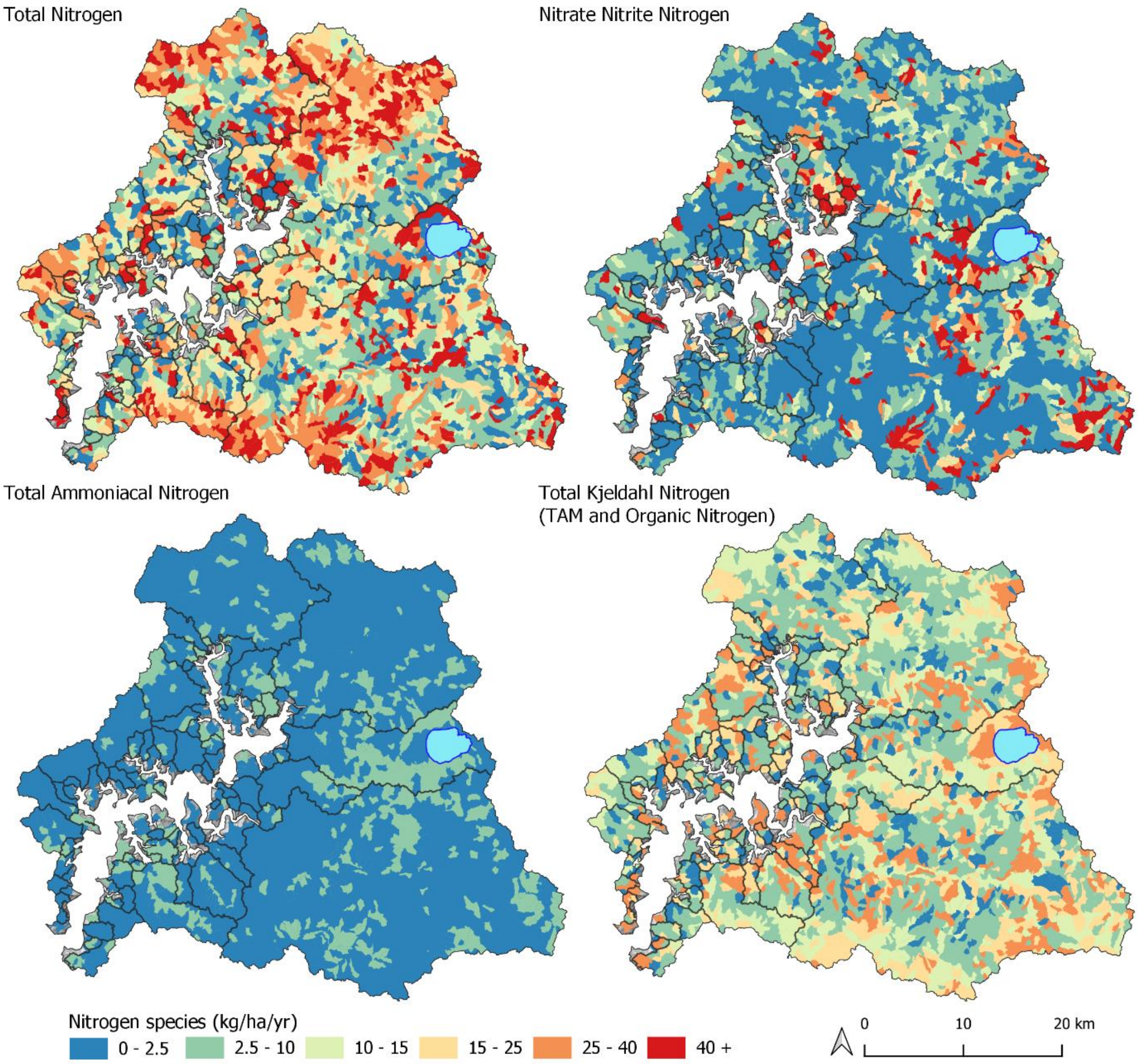


Figure 8. Predicted nitrogen export load to the Hokianga Harbour in kg ha^{-1} yr^{-1} over REC1 stream orders 1 to 7. TN is the sum of NNN and TKN. TKN includes both ammoniacal (TAM) and organic nitrogen.

Total phosphorus (TP) includes particulate and dissolved forms of P (Figure 9). Dissolved reactive phosphorus (DRP) exhibits a strong landscape control, primarily related to the P-rich basalts of the Tangihua Volcanics and a lesser degree peat wetland and organic soils (Figure 9). The importance of basalt over P-export was noted in the steady-state model and continues to be an important predictor of DRP load (Rissmann and Pearson, 2020). Commonly, particulate phosphorus (PP) makes up a significant component of the TP lost from agricultural land, with the export coefficient map suggesting a significant PP component from poorly drained soils associated with intensive land use. Here, surface runoff is likely an important source of PP. However, an association with wetlands and organic soils is also noted. Organic-P in dissolved or even colloidal form may be an important component of P-export to stream from wetlands and/or reducing aquifer systems.

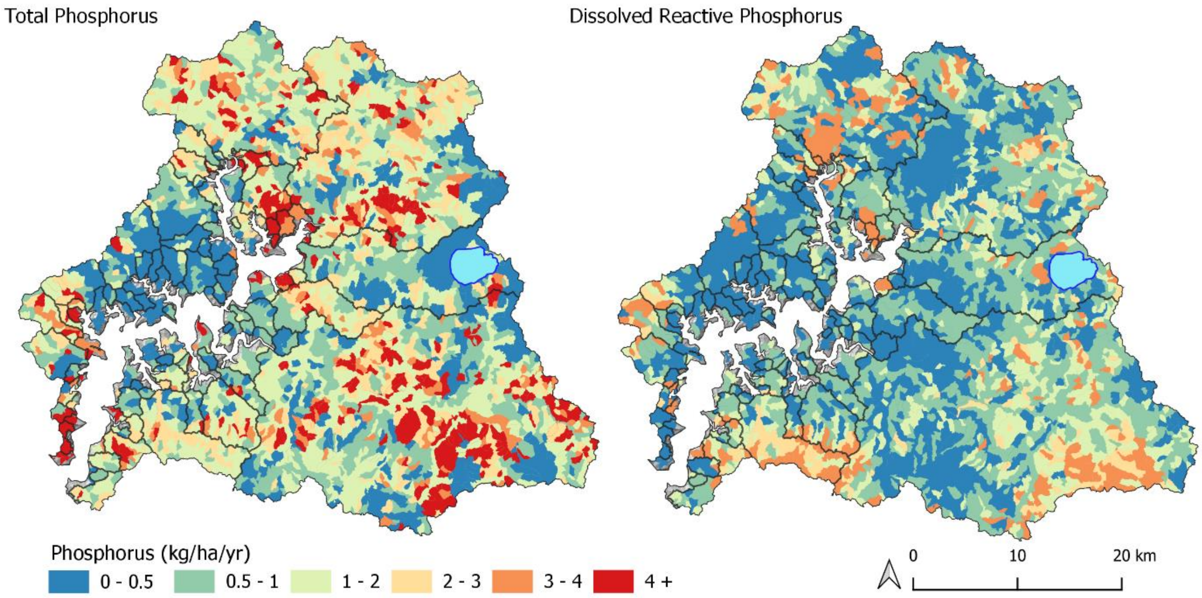


Figure 9. Predicted phosphorus export load to the Hokianga harbour in kg ha^{-1} yr^{-1} over REC1 stream orders 1 to 7. TP is the measure of both particulate phosphorus and DRP.

Total suspended sediment (TSS) has the fewest number of sites to generate a regional model and is associated with the greatest uncertainty of all water quality models. Current work by Northland Regional Council is looking to increase the number and representativeness of TSS measures across the Northland Region. Until this work is done, it is difficult to have much confidence in the resultant export coefficients with greater certainty associated with the high-resolution depiction of erosion susceptibility and subsequent validation through satellite and ground-based observation. Despite the limitations, TSS appears to have a strong land use control with the more stable landscapes relatively undisturbed in the natural state forest (Figure 10).

E. coli export is also highly uncertain, due mainly to the complexity of loading, source build up and depletion, and other complex behaviours (Figure 10). However, a reasonable spatial correlation with land use intensity is observed.

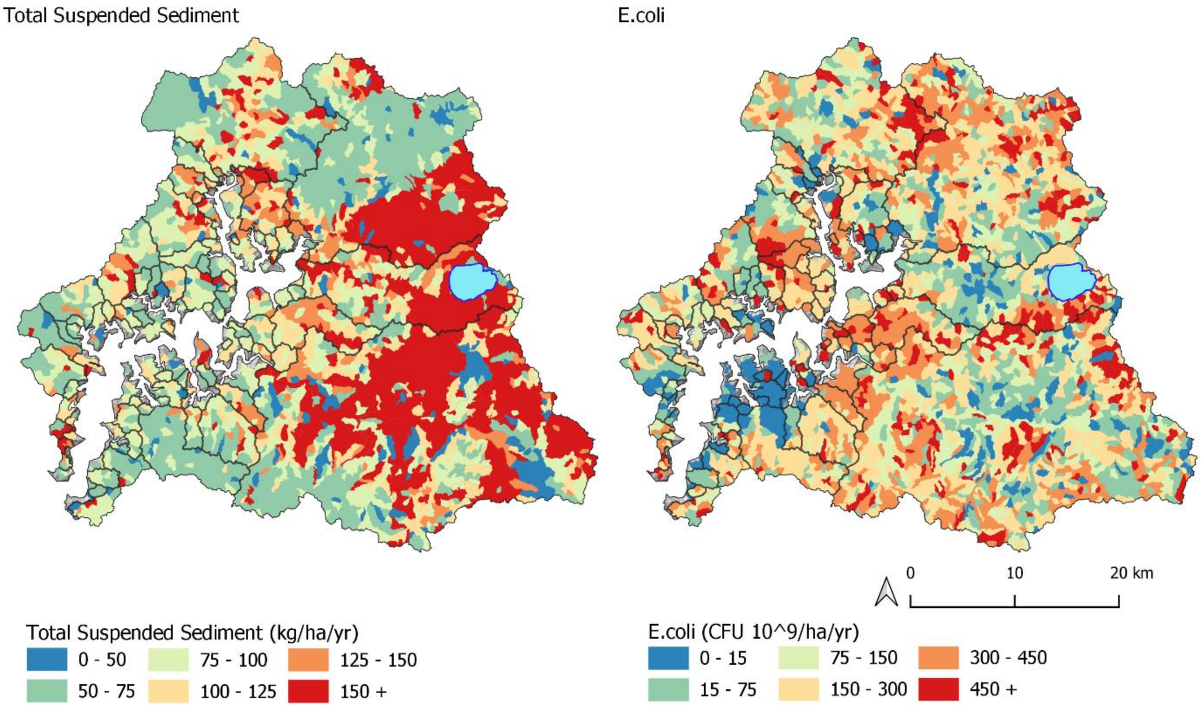


Figure 10. Predicted total suspended sediment export load in kg ha^{-1} yr^{-1} and E.coli in CFU 10^9 ha^{-1} yr^{-1} to the Hokianga harbour over REC1 stream orders 1 to 7.

A summary of the subcatchments for each contaminant, TN, TP, TSS, and *E.coli*, is presented as an export load in Figure 11. The discussion above for each analyte is also relevant at the subcatchment scale. The larger areas for model application reduce some of the limitations apparent above. For this reason, order 1 streams and the zone of direct discharge have been calculated together.

Estimated annual load for the Hokianga Harbour is calculated from the sum of the subcatchment export yield multiplied by the catchment area (Ha). The total load to the Hokianga harbour is estimated at 1,677 tonnes of nitrogen, 166 tonnes of phosphorus, 19,782 tonnes of sediment, and 15,542,430 CFU 10^9 for *E.coli* (Table 16). The contribution of NNN and DRP is provided for assessment of estuary trophic index (ETI, Section 6). The contribution to TN and TP from NNN and DRP is 28% and 42%, respectively.

Table 16. Modelled load estimate for the Hokianga harbour.

	TN	NNN	TP	DRP	TSS	E.coli (CFU10^9)
Load estimate (T/yr)	1,677.19	466.20	165.99	69.17	19,781.71	15,542,429.57
Export coefficient (kg/ha/yr)	10.55	2.93	1.04	0.44	124.49	97.81
95% confidence interval	154.54	86.24	13.73	5.28	1,905.67	1,596,842.63
Upper load estimate (T/yr)	1,831.73	552.45	179.72	74.45	21,687.38	17,139,272.21
Lower load estimate (T/yr)	1,522.65	379.96	152.26	63.89	17,876.03	13,945,586.94

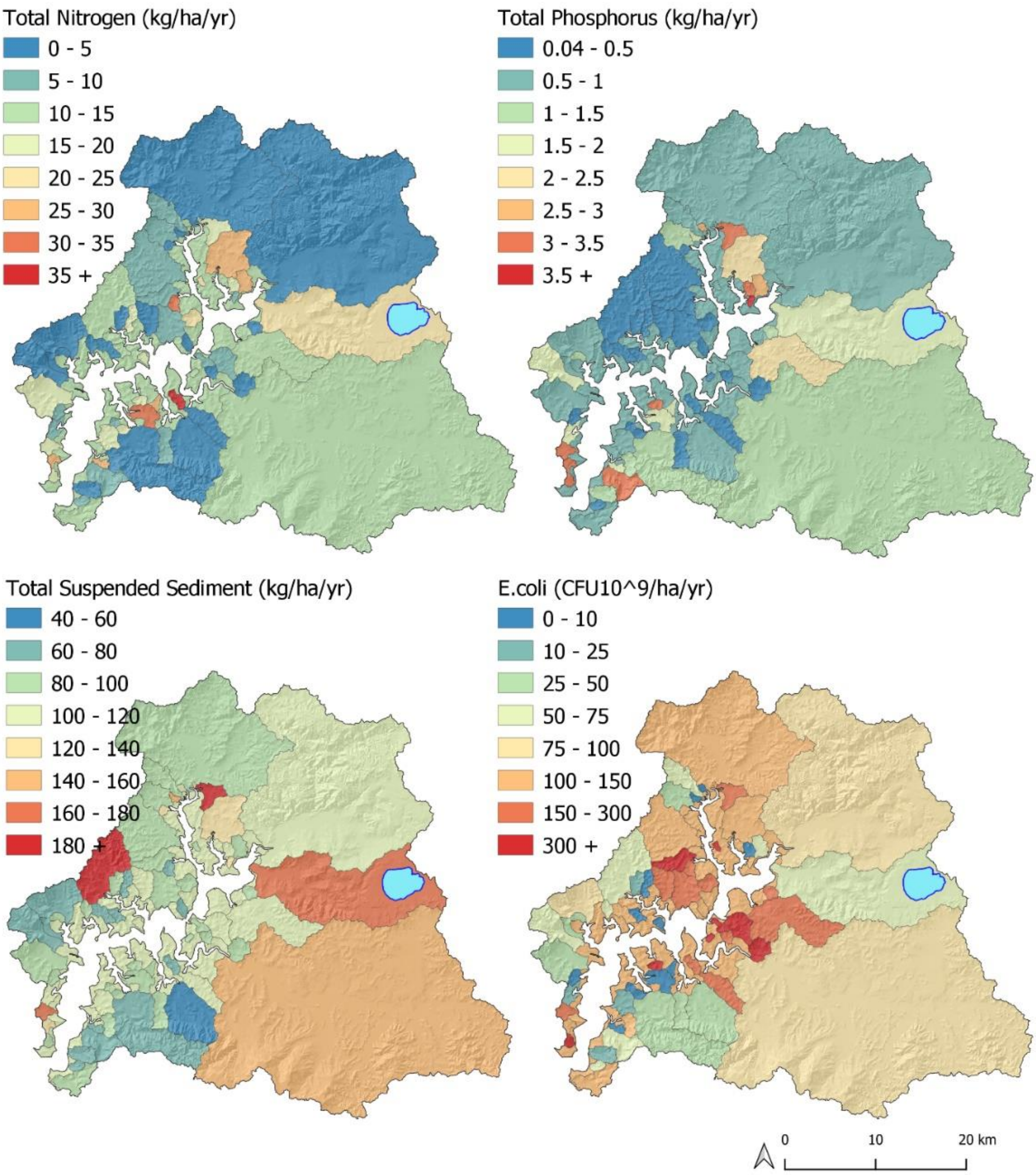


Figure 11. Predicted export load for total nitrogen (top left), total phosphorus (top right), total suspended sediment (bottom left) in kg ha^{-1} yr^{-1} and E.coli in CFU 109 ha^{-1} yr^{-1} (bottom right) for the Hokianga Harbour subcatchments.

5.3 Point source discharges

5.3.1 Farm Dairy Effluent

Most animal effluent (urine and dung) is deposited directly onto pasture as a diffuse discharge. The typical concentration of effluent derived from several different research trials is estimated to be 269 g N m^{-3} (range 181 – 506) and 69 g P m^{-3} (range 21-82) (Longhurst et al. 2000, Houlbrooke, 2008). However, approximately 6-10 % of effluent is deposited in the milking shed and collecting yards. When the yards and milking area are cleaned after milking, farm dairy effluent (FDE) is generated at approximately 50 L per cow per day (Selvarajah, 1999; Bolan et al., 2009). FDE is collected and stored in ponds.

FDE is commonly treated using a two-pond system combining both an anaerobic and facultative pond (Sukias et al. 2001). The combination of an anaerobic and aerobic pond efficiently removes sediment and biological oxygen demand (BOD). Pond systems, however, are not primarily designed to remove nutrients, such as nitrogen and phosphorus, or microbes resulting in high concentrations in the FDE (Longhurst et al. 2000; Sukias et al. 2001; Craggs et al. 2003). Longhurst et al. (2000) reported that effluent discharging from a standard two-pond system to surface waters has mean concentrations of approximately 91 g N/m^{-3} and 23 g P/m^{-3} (Houlbrooke, 2008). The two-pond treated effluent represents a significant improvement when compared to the nutrient concentrations of raw FDE, however, the concentrations of N and P in discharges from a two-pond treatment system are still more than three orders of magnitude greater than levels considered likely to promote aquatic weed growth (0.61 g N m^{-3} and 0.033 g P m^{-3} respectively) (ANZECC, 2000). Land application of FDE became the preferred treatment option for many Regional Councils in the 1990s, allowing the water and nutrients applied to land in FDE to be utilised by the soil-plant system. In Northland, FDE can be discharged to either land or water.

To estimate the load to the Hokianga harbour from FDE discharge, information was provided by NRC for analysis (Rachael Anderson, February 2020[1]). This included the location of the discharge, the number of cows on each farm, the discharge type, number of ponds, and the milking regime. FDE discharges for Northland farms are classified as either a permitted activity or a discharge permit. To meet permitted activity status, all FDE must be discharged to land. For those holders of discharge permits, FDE is either discharged to water (Land application = no) or disposed to land unless weather and other conditions are not suitable, in which case they discharge from the pond system to surface water (Land application = yes). The milking regime is either twice a day or once a day milking. For once a day milking, NRC staff estimate a 30% reduction in FDE generated, or 35 L per cow per day (Rachael Anderson, Farm Monitoring Manager). The milking season typically occurs over a 9-month period (274 days). This information was used, in conjunction with published literature, to estimate the contaminant load from FDE ponds to the Hokianga Harbour.

A budget was calculated for the Hokianga harbour catchment to estimate the likely contaminant load from FDE. The following assumptions were made regarding nutrient and microbial concentrations and volumes:

- The nutrient composition of farms FDE with a single pond is assumed to be similar to effluent (269 g N/m^{-3} and 69 g P/m^{-3}; Longhurst et al., 2000).
- FDE discharged from a two-pond system assumes a reduction in nutrient concentration to 91 g N/m^{-3} and 23 g P/m^{-3} (Houlbrooke, 2008).
- Suspended solid concentrations is assumed to be 206 g m^{-3} (Sukias, et al., 2001)
- Microbial concentration is assumed to be 24,000 CFU per 100 ml (24 CFU 10^9) (Sukias, et al., 2001; Donnison et al., 2011; Palliser et al., 2015)
- The daily volume of FDE produced is 0.05 m^3 per cow in traditional twice a day milking systems and 0.035 m^3 for once a day milking.
- The milking season is 274 days long. For farms that operate a split calving season, estimates were based on total cow numbers assuming the same length milking season regardless of when milking occurred.

[1] This information was up to date at the time of provision. It is noted that dairy supply numbers have changed over the 2015-2019 model calibration period and are indicative only.

- FDE from permitted activity status or discharge permit with land application is 100% discharged to land. FDE from discharge permit with no land application is 100% discharged to water.

Table 17 shows the load to the Hokianga Harbour from the different FDE discharge types. Permitted activity status and discharge permit with land application is considered diffuse discharges and discharge permit to water is a point source discharge. The total FDE catchment load is unlikely to be realised due to nutrient uptake and attenuation from the land. Howard-Williams et al. (2010) estimate the percentage of the TN load lost from dairy and other pasture to be 36.7% and 33.3%, respectively, with the remainder attenuated within the catchment. Based on this evidence, FDE discharges to land are likely to account for approximately 7.17 T/yr.

FDE nutrient and *E.coli* concentrations taken from the literature are the largest limitations in this assessment as the remainder of the data came from NRC records. The accuracy of these estimations can be validated by the sampling of a number of dairy effluent ponds to produce a better estimate of FDE composition for Hokianga dairy farms.

Table 17. Nutrient and microbial load from FDE discharges.

	TN (T/yr)	TP (T/yr)	TSS (T/yr)	*E.coli* (CFU $10^{9)}$
Discharge Permit – to water	5.18	1.31	11.72	1,365,178
Discharge Permit – to land	11.30	2.86	25.58	2,980,243
Permitted activity – to land	8.24	19.75	10.19	1,187,461
Total FDE	24.71	23.91	47.49	5,532,882
% FDE load to catchment from discharges to water	0.31	0.79	0.06	8.78
% total FDE catchment load	1.47	14.41	0.24	35.60

5.3.2 Wastewater Treatment Plants

There are four wastewater treatment plants (WWTP) in the Hokianga Harbour catchment. The largest discharge is from Kaikohe WWTP which is located in the upper Waima catchment, up catchment of the Punakaitere at Takehe monitoring site. The other three discharges are small direct discharge to the harbour (Figure 3). Loads were calculated from the available data at the discharge to the constructed wetlands using the same methods applied to the water quality monitoring sites (Table 18). The summary statistics of the load estimates provides an integrated measure of likely load that has the benefit of a confidence interval and associated range for each site (Table 19). Outliers were removed from the calculation of mean scores and associated descriptive statistics and confidence intervals (95%).

Table 18. Load estimate for WWTP sites in tonnes per year and CFU 10^9 per year for E.coli from all methods. The cluster membership of the load equations is indicated by shading for each method (yellow – largest cluster, blue – 2nd cluster and orange – 3rd cluster). Outliers identified by the clustering are indicated in green. The methods of Fraser and Snelder (2020) were unable to be applied to E.coli loads.

T/yr	Method 1	Method 2	Method 3	Method 4	Method 5	Method 6	Beale.ratio	Rating	Ferguson	L7	L5	FS	FST
Kaikohe													
TN	19.009	18.367	23.122	24.010	23.199	22.731	23.190	19.583	26.347	178.993	186.635	12.376	22.595
NH_4	16.061	15.058	17.479	20.286	19.020	19.151	19.011	16.663	17.874	17.042	17.747	10.472	19.970
DIN	17.411	16.650	20.208	21.992	21.030	20.647	21.024	18.188	19.876	18.938	19.478	11.435	21.413
TP	2.801	2.586	3.343	3.538	3.266	3.187	3.264	2.873	3.070	2.919	3.079	1.815	3.454
DRP	1.916	1.732	2.415	2.421	2.187	2.133	2.185	1.850	2.029	1.967	2.043	1.238	2.295
TSS	35.506	34.025	45.103	44.847	42.976	40.889	42.936	33.473	44.081	40.214	47.542	22.396	37.047
E.coli	70542.41	61997.84	66224.49	77493.47	68106.92	86611.78	67734.26	27414.55	82159.85				
Kohukohu													
NH_4	0.326	0.329	0.291	0.294	0.296	0.290	0.296	0.279	0.298	0.318	0.320	0.153	0.303
TSS	0.249	0.227	0.190	0.224	0.205	0.185	0.204	0.138	0.200	0.224	0.209	0.112	0.173
E.coli	1258.262	1334.707	999.312	1173.267	1244.548	900.680	1241.635	220.328	1456.851				
Opononi													
NH_4	3.426	3.201	3.041	2.041	1.907	2.403	1.904	1.679	2.021	1.169	1.462	1.026	1.265
TSS	2.702	2.579	2.336	1.610	1.537	1.796	1.536	1.115	1.516	12.812	2.230	0.774	1.729
E.coli	7570.71	9543.82	7864.35	7430.59	9367.19	5254.93	9392.18	3349.17	10981.91				
Rawene													
TN	1.463	1.483	1.666	1.708	1.732	1.695	1.733	1.651	1.718	1.880	1.750	0.849	1.650
NH_4	1.514	1.522	1.688	1.769	1.778	1.764	1.778	1.600	1.743	1.701	1.693	0.844	1.553
TP	0.211	0.204	0.237	0.246	0.238	0.227	0.238	0.209	0.230	0.229	0.235	0.119	0.223
TSS	0.767	0.744	0.845	0.895	0.869	0.813	0.868	0.689	0.872	0.747	0.783	0.405	0.631
E.coli	413.67	405.67	469.33	479.39	470.12	489.66	469.34	174.34	486.09				

Table 19. Load statistics (median, mean, minimum, maximum, range, 95% confidence interval, standard deviation, and coefficient of variation) at WWTP sites. Median is calculated from all available load methods. The method identifies the load calculation method, which corresponds to the median value. No of methods is the count of methods used to calculate the summary statistics (except for median values).

	Median	Method	Mean	Minimum	Maximum	Range	95% C.I.	S.D.	C.V.	No. Methods	1	2	3
Kaikohe													
TN	23.122	Method 3	46.17	12.38	186.63	174.26	35.28	58.38	1.26	13	21.32	182.81	
NH_4	17.747	L5	17.95	15.06	20.29	5.23	0.97	1.53	0.09	12	16.85	19.49	
DIN	19.876	Ferguson	19.74	16.65	21.99	5.34	1.01	1.59	0.08	12	18.68	21.22	
TP	3.079	L5	3.02	1.82	3.54	1.72	0.26	0.43	0.14	13	3.28	2.60	
DRP	2.043	L5	2.03	1.24	2.42	1.18	0.18	0.30	0.15	13	1.82	2.27	
TSS	40.889	Method 6	39.31	22.40	47.54	25.15	3.94	6.52	0.17	13	43.57	32.49	
E.coli	68106.92	Method 5	67587.29	27414.55	86611.78	59197.23	12363.63	16084.49	0.24	9	66921.18	82088.37	
Kohukohu													
NH_4	0.296	Method 5	0.30	0.28	0.33	0.05	0.01	0.02	0.05	12	0.29	0.32	
TSS	0.204	Beale.ratio	0.20	0.11	0.25	0.14	0.02	0.04	0.18	13	0.19	0.23	0.12
E.coli	1241.63	Beale.ratio	1201.16	900.68	1456.85	556.17	139.21	166.51	0.14	8	1284.88	950.00	
Opononi													
NH_4	1.907	Method 5	2.04	1.03	3.43	2.40	0.45	0.75	0.37	13	2.06	1.32	3.22
TSS	1.729	FST	1.79	0.77	2.70	1.93	0.35	0.55	0.31	12	1.45	2.46	
E.coli	7864.35	Method 3	7861.65	3349.17	10981.91	7632.74	1711.19	2226.18	0.28	9	8878.68	4302.05	
Rawene													
TN	1.695	Method 6	1.68	1.46	1.88	0.42	0.07	0.11	0.06	12	1.72	1.47	
NH_4	1.693	L5	1.68	1.51	1.78	0.26	0.06	0.10	0.06	12	1.74	1.55	
TP	0.229	L7	0.23	0.20	0.25	0.04	0.01	0.01	0.06	12	0.23	0.21	
TSS	0.783	L5	0.79	0.63	0.90	0.26	0.05	0.08	0.10	12	0.86	0.73	
E.coli	469.34	Beale.ratio	428.62	174.34	489.66	315.32	72.50	94.31	0.22	9	477.32	331.23	

Table 20 provides a summary of the wastewater treatment plants discharge to constructed wetlands. It is likely there is further attenuation of contaminants occurring in the wetland prior to reaching the harbour.

Table 20. Summary load from WWTP to the Hokianga Harbour for TN, NNN, TP, DRP, TSS, and E.coli.

	TN (T/yr)	NNN (T/yr)	TP (T/yr)	DRP (T/yr)	TSS (T/yr)	E.coli (CFU10^9/yr)
Kaikohe	23.12	2.13	3.08	2.04	40.89	68106.92
Kohukohu	0.30				0.20	1241.63
Opononi	1.91				1.73	7864.35
Rawene	1.69		0.23		1.69	469.34
WWTP Total	27.02	5.38*	3.31	2.04*	43.61	77682.24
% catchment load	1.61	1.15*	1.99	2.95*	0.22	0.50

Note: Not all analyte data is measured at all WWTP discharges. For Kohukohu and Opononi, WWTP data is collected for NH_4, TSS, and E.coli only. For Kaikohe, NNN is calculated from the difference between DIN and NH_4 (Table 18).
*Does not include load from Kokukohu, Opononi and Rawene.

5.3.3 Other discharges

Information to support the calculation of contaminant load is limited for other point source discharges, the majority of which are stormwater discharges (Figure 3). These discharges have been assessed by NRC staff and deemed to be of 'low risk' and therefore not a significant contributor to catchment load. Inherently many of these discharges are likely to be accounted for in the load estimates and subsequent models as they are upstream of State of Environment Water Quality Monitoring sites.

5.4 Historic Contaminant Load

Historic or pre-human loads to the catchment were estimated from the export coefficient from current monitoring sites with associated flow data in the Northland region that have greater than 80% natural state land cover (Pearson et al., 2020). There are three sites in the region that meet these criteria, Waipapa at Forest Ranger, Victoria at Victoria Valley Road, and Waipoua at SH12. The export coefficients for each contaminant are presented in Table 21.

Table 21. Percentage of water quality site capture area in natural state and predicted export load for TN, TAM, TKN, NNN, TP, DRP, TSS in kg ha^{-1} yr^{-1} and microbial load as indicated by E.coli in CFU 10^9 ha^{-1} yr^{-1}.

Site Name	% NS	TN	TAM	TKN	NNN	TP	DRP	TSS	*E.coli*
Waipapa at Forest Ranger	86.63	7.13	0.23	-	1.88	0.62	0.25	-	53.65
Waipoua at SH12	93.80	8.66	0.37	8.24	1.83	0.69	0.35	58.43	35.08
Victoria at Victoria Valley Road	83.69	6.62	0.38	6.71	0.65	1.4	1.02	73.62	36.66
Average		7.47	0.33	7.48	1.45	0.90	0.54	66.03	41.80
Median		7.13	0.37	7.48	1.83	0.69	0.35	66.03	36.66

Average export coefficients for TN, TSS, and *E.coli* were multiplied with the catchment area to estimate the historic load to the catchment (Table 22). TP was calculated from the sum of the estimated load from the area of Tangihua basalt, which is naturally higher in phosphorus (represented by the Victoria at Victoria Valley Road site at 1.4 kg/ha/yr), and the remainder of the catchment (average export coefficient of 0.65 kg/ha/yr) (Table 20). Comparison of the estimated pre-human load with the current load indicates the increase to the Hokianga Harbour over the historic load is estimated to be approximately 30% for TN, 28% for TP, 48% for TSS, and 58% for *E.coli* (Table 22). There is a lesser degree of certainty around the historic load for sediment and microbes as modelled load could be highly variable between the 13 methods applied (Pearson et al., 2020). TSS is also strongly correlated with the Erosion Susceptibility Classification of Rissmann et al. (2018b) and areas remaining in natural state land cover are those that have been identified as typically highly erodable steeper terrain or elevated plateaus of the Tangihua Basalts. Therefore, the TSS export coefficients calculated for these areas may overestimate the load for those more stable landscapes that are now under agricultural land use.

Table 22. Pre-human and current load to the Hokianga Harbour. TN, TP, and TSS is in tonnes per year and microbial load as indicated by E.coli in CFU 10^9 per year.

	TN	TP	TSS	*E.coli*
Pre-human load	1,169.38	119.46	10,331.84	6,543,497
Current load	1,677.19	165.99	19,781.71	15,542,430
Anthropogenic increase over pre-human load (%)	30.3	28.0	47.8	57.9

Snelder et al. (2017) estimated the anthropogenic increases of catchment nutrient loads across New Zealand. The method developed a spatial regression model to predict export coefficients (kg ha^{-1} yr^{-1}) for nitrogen and phosphorus under natural and current conditions. The models were derived using loads (kg yr^{-1}) of TN, NO_3-N, TP, and DRP calculated for 592 river water quality monitoring sites nationally. Snelder et al., (2017) found that anthropogenic increases in nutrient export above natural levels were associated with the proportions of catchments occupied by intensive agricultural land cover and anthropogenic increases varied significantly between regions.

The model data was obtained from T. Snelder (August 2020) and analysed for the Hokianga catchment for comparison with this work (Table 23). The anthropogenic increase over natural state to the Hokianga Harbour was significantly lower than the national average for TN and most significantly for TP. Overall, estimates of current and historic catchment load from Snelder et al. (2017) were lower than those calculated in Table 22. However, it is important to note the model of Snelder et al. (2017) is unlikely to account for the role of geochemical variation over catchment load, for both current and historic estimates.

Table 23. Pre-human and current load (T/yr) to the Hokianga Harbour from Snelder et al., (2017).

	TN	TP
Pre-human load	747.86	82.96
Current load	1240.60	87.54
Anthropogenic increase in load to the Hokianga Harbour (%)	39.7	5.2
National anthropogenic increase in load (%)	74	48

5.5 Catchment Budget

A catchment budget was constructed from the load estimates for TN, NNN, TP, DRP, TSS, and *E.coli* (Table 24). The natural state diffuse load was estimated using the same method as the historic contaminant load where the area of natural state land cover was multiplied by the average export coefficient from the three natural state reference sites (Table 21). The export coefficients used to calculate TP and DRP were proportional to the area of Tangihua Basalts. The anthropogenic diffuse load was estimated from the total catchment load minus the natural state and point source discharges.

Significant land use change has occurred in the catchment since pre-human state with only 45% of the natural land cover remaining in the catchment. Natural state contaminant load is now a minor component of the total contaminant load, with the exception of DRP (Table 24). Currently, contaminant load from agricultural land use and forestry account for 68% of the TN load, 65% of the TP load, 76% of the TSS load, and 81% of the microbial load as indicated by *E.coli*.

Table 24. Hokianga Harbour catchment budget in T/year for TN, NNN, TP, DRP, and TSS and CFU 10^9 for E.coli.

	Area (Ha)	TN	NNN	TP	DRP	TSS	*E.*
Diffuse sources							
Land in natural state	72,404.0	540.86	132.50	58.45	32.73	4,780.84	3,026,487
Agricultural and forestry land use	86,212.0	1,104.13	328.32	102.92	34.40	14,945.54	11,073,082.13
Diffuse Source Total		1,644.99	460.82	161.37	67.13	19,726.38	14,099,569
Point sources							
WWTP		27.02	5.38	3.31	2.04	43.61	77,682
FDE		5.18		1.31		11.72	1,365,178
Point Source Total		32.20	5.38	4.62	2.04	55.33	1,442,860.24
Total catchment load		1,677.19	466.20	165.99	69.17	19,781.71	15,542,430
% Load from land in natural state		32.25	28.42	35.21	47.31	24.17	19.47
% Load from agricultural land and anthropogenic sources		67.75	71.58	64.79	52.69	75.83	80.53

Note: NNN and DRP loads from FDE were not calculated.

The catchment budget was compared with a load estimate calculated from export coefficients derived by land use for the harbour (Table 25; Quinn et al., 2009; McDowell and Houlbrooke, 2008). Comparison with published literature for nutrients show a similar prediction for TN load. It appears that TP is underestimated by export coefficients from the published literature likely due to the inability to account for the geological contribution to the contaminant load. The value predicted is similar to that of Snelder et al. (2017).

Table 25. Nutrient budget estimated from literature export coefficients by land use.

Land use	Area (ha)	N (kg/ha/yr)	TN Load (T/yr)	P (kg/ha/yr)	TP Load (T/yr)
Diffuse Sources					
Native forest	70,541	5	352.71	0.2	14.11
Plantation forest	21,620	5	108.10	0.2	4.32
Grassland – Dry stock	52,802	12	633.62	0.8	42.24
Grassland - Dairy	11,016	39	429.62	1.2	13.22
Cropland	198	30	5.94	0.8	0.16
Other	366	5	1.83	0.1	0.04
Diffuse Source Total	156,543		1,531.82		74.09
Point Sources					
WWTP			27.02		3.31
FDE - to water			5.18		1.31
Point Source Discharge Total			32.2		4.62
Total Load			1,564.02		78.71

6 Estuary Trophic Index

This section was provided by Northland Regional Council's Coastal Resource Scientist Richard Griffiths.

The Estuary Trophic Index (ETI) was developed by Wriggle Limited and NIWA to assist Councils to determine the susceptibility of an estuary to eutrophication, assess its current trophic state, and assess how changes to nutrient loads may alter its current course (Roberston *et al.* 2016; Zeldis et al., 2017). This assessment utilised the Screening Tool 1 (https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/). Screening Tool 1 provides an assessment of the susceptibility of the system to eutrophication based on the estuaries physical characteristics and the nutrient load to estuary response relationships for specific estuary types. The tool reports a physical susceptibility (very high, high, moderate, low susceptibility) and a combined physical and nutrient load susceptibility rating (Band A (no stress) – Band D (significant persistent stress)).

The medians from the 13 different load calculation methods were used as the input values for the annual river TN loading and the annual river TN loading, required by the ETI tool (Table 26). The upper and lower 95th percentile, and the pre-human estimates of the annual river TN loading, and the annual river TP loading were also utilised for separate calculations of susceptibility (Table 26).

Table 26. Input river nutrient loading values used to calculate susceptibility for ETI tool 1.

	Annual river total nitrogen loading (T/yr)	Annual river total phosphorus loading (T/yr)
Median load (from Table 16)	1,677.19	165.99
Upper load (from Table 16)	1,831.73	179.72
Lower load (from Table 16)	1,522.65	152.26
Pre human load (from Table 22)	1,169.38	119.46

The default values provided with the ETI tool 1 were used for the other input values (Table 27). A separate calculation of susceptibility was also undertaken using the median load estimate (from Table 26) and an updated NO_3-N/TN ratio of 0.28 and the DRP/TP ratio of 0.42, based on the modelled load estimates in this report (Table 16), instead of the default ETI ratios of 0.75 and 0.72 respectively.

Table 27. ETI tool 1 input values.

ETI Tool 1 input parameters	ETI default values
ETI classification	Shallow intertidally dominated estuary
Freshwater inflow	43.20803 m^3/s
Volume	4.83E+08 m^3
Tidal prism	2.16E+08 m^3
Ratio NO_3	0.75039
Ratio DRP	0.724415
Ocean salinity	35.29808
Ocean nitrate concentration	35.55011 mg/m^3
Ocean DRP concentration	9.386513 mg/m^3
Intertidal area	48.69%
Estuary area	1.06E+08 m^2
Mean depth	4.535039 m
Tidal height	2.683 m

The output from the ETI tool calculator indicates that the physical susceptibility of the estuary is low, and that the nitrogen load susceptibility is moderate (Table 28). The ETI susceptibility, which takes into account the physical characteristics and the nutrient load to estuary response relationship, was band B, with a macroalgae susceptibility of band B and a phytoplankton band B, for all of the input load values (Table 28). The following guidance is provided by Robertson et al. (2016) for shallow intertidally dominated estuaries within susceptibility band B. "*A minor stress on sensitive biota caused by the indicator. Some eutrophic symptoms (e.g. macroalgae) but still [able to] support healthy seagrass and fish communities*".

Although there are no differences in the ETI susceptibility bands with different input load values, the ETI output indicates that there are differences in the estuarine nutrient concentrations (Table 29). The estuarine TN and TP concentrations increase from 0.097 mg/l and 0.015 mg/l respectively from the historic pre-human river input loads to 0.125 mg/l and 0.018 mg/l with the current input loads (Table 29). The estuarine chlorophyll-a concentration also increases from 0.003 mg/l to 0.007 mg/l.

Table 28. ETI Tool 1 results for the current load.

	ETI Susceptibility	Macroalgae Band	Phytoplankton Band	Physical Susceptibility	N Load Susceptibility
Current median	B	B	B	Low	Moderate
Upper 95th confidence interval	B	B	B	Low	Moderate
Lower 95th confidence interval	B	B	B	Low	Moderate
Current median (updated NO_3 and DRP ratios	B	B	B	Low	Moderate
Historic (Pre-human) load	B	B	B	Low	Moderate

Table 29. ETI Tool 1 concentrations for the current load.

	Estuary TN (mg/l)	Estuary TP (mg/l)	Estuary NO_3 (mg/l)	Estuary DRP (mg/l)	Estuary Chl-a (mg/l)
Current median	0.1248	0.0178	0.1019	0.0153	0.0066
Upper 95th confidence interval	0.1333	0.0185	0.1083	0.0158	0.0076
Lower 95th confidence interval	0.1164	0.0170	0.0955	0.0147	0.0057
Current median (updated NO_3 and DRP ratios	0.1248	0.0178	0.0586	0.0125	0.0066
Historic (Pre-human) load	0.0970	0.0152	0.0810	0.0134	0.0034

7 Limitations and Recommendations

Load estimates based on monthly water quality sampling are inherently uncertain. As discussed by Snelder et al. (2014) the loads calculated from monthly monitoring data are commonly associated with significant uncertainties because sampling at such low frequency fails to represent the population of daily loads adequately (Defew et al., 2013; Robertson and Roerish, 1999; Johnes, 2007; Philips et al., 1999). Snelder et al. (2014) cite the works that calculate errors of between 200 – 500%. There is no reason to expect that the estimates for the Northland region are any less certain, especially for TSS, for which the number of samples available for load estimation is limited.

For pragmatic reasons, current monitoring and flow sites are biased towards higher-order streams. Models developed on this data, therefore, may not be representative of smaller order capture zones and associated 1st and 2nd order streams or areas outside the calibration extent, especially small sea draining catchments. The amplification of error at small scales is well recognised. The latter in conjunction with the small sample size, regionally n = 31 sites, means that export coefficients should not be over-interpreted but used within the context of other relevant information. For example, ground-based evidence for erosion susceptibility and high-intensity land use.

Of the 67 surface water quality monitoring sites within the region, only 31 have flow measurements suitable for load estimation. Measuring flow at a larger number of water quality monitoring sites would improve the value of the data collected in the region. Due to these limitations, the steady-state modelling is considered to provide a more robust representation of landscape-based controls over water quality. This reflects the greater number of sites used for model development, and the use of median values for model development. Limitations associated with physiographic layers and the modelling approach applied here are discussed in detail in Rissmann and Pearson (2020).

8 Summary

Water pollution, now overwhelmingly from diffuse sources, has been well documented and the management of diffuse pollutants is currently receiving considerable attention nationally through the development and implementation of the National Policy Statement for Freshwater Management (Land and Water Forum, 2010, Ministry for the Environment, 2009; 2014; 2017; 2020). Given the large area of pastoral farming, it is not surprising that New Zealand suffers considerable diffuse water pollution, and the link between pastoral land use intensification and declining water quality is well recognised (Ministry for the Environment, 2009; Parliamentary Commissioner for the Environment, 2012; Snelder et al., 2017).

Overall, diffuse sources to the Hokianga harbour make up the largest contribution of contaminant load. Total nitrogen load has increased by approximately 34% due to anthropogenic derived sources and now represents 68% of the total input. Due to the landscape characteristics, the dominant form of nitrogen is organic, which accounts for approximately two-thirds of the total nitrogen load. Total phosphorus load has increased by approximately 28% due to anthropogenic derived sources and now represents 65% of the total input. The main form of phosphorus load to the harbour is from particulate phosphorus. Weathering processes and sediment erosion contribute approximately half of the dissolved reactive phosphorus from predominantly natural state areas due to the phosphorus rich Tangihua Basalts.

Physiographic information can be used to identify the dominant pressure on water quality in the Hokianga Harbour catchment (i.e. landscape erosion, land use, or both) (Figure 12). The blue areas represent where pressure from the landscape exhibits a larger control over the contaminant load than land use, these areas are predominantly under natural forest land use. Yellow areas represent equal pressure and can either be high or low risk. Red areas are where the land use pressure is greater than the erosion risk.

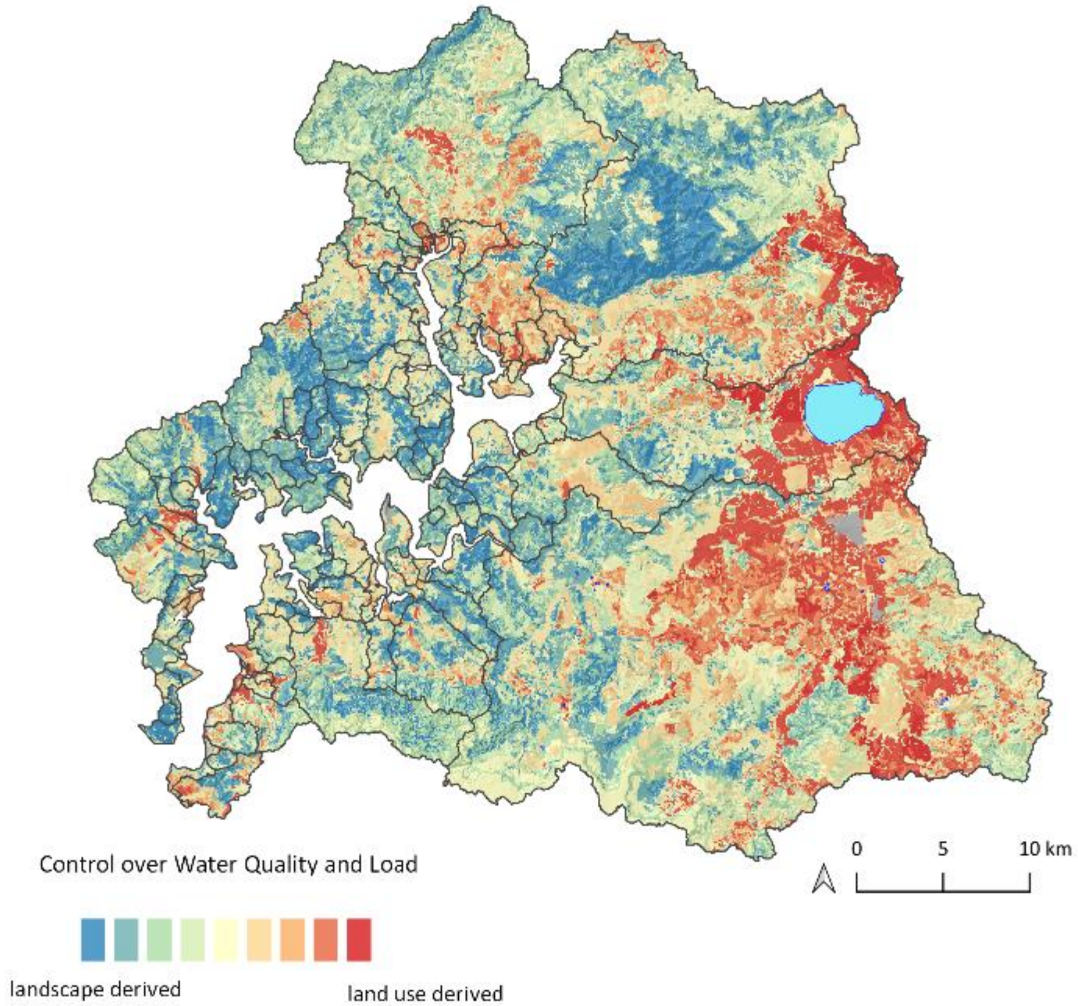


Figure 12. Risk to water quality from erosion (particulate transport) and land use pressures. Where blue areas represent where pressure from the landscape exhibits a larger control over the contaminant load than land use, yellow areas represent equal pressure and can either be high or low risk and red areas are where the land use pressure is greater than the erosion risk.Mapped from the erosion susceptibility classification of Rissmann et al. (2018b) and the physiographic land intensity layer (ESC – LUI).

The sediment load to the harbour has nearly doubled (48% increase) since pre-human times. Currently, anthropogenic derived sources (predominantly agricultural and forestry) account for 76% of the current total load to the Hokianga Harbour. Figure 13 combines the risk to water quality from landscape (erosion risk and particulate transport) and land use pressures. The areas identified as high (red) are where landscape susceptibility and land use pressure combine to generate a high-risk of sediment loss. These areas are also likely to be contributing a disproportionately high microbial load to the harbour as surficial runoff is a common occurrence in poorly drained heavy clay soils. It is estimated that 80% of the microbial load is from diffuse agricultural land use sources, an increase of approximately 64% from pre-human conditions.

Water quality issues in the Hokianga Harbour, such as excessive siltation and microbial pollution, have been associated with the increased surficial runoff from pastoral land and accelerated sediment deposition since European colonisation (Morrison, 2005; Woods and Kennedy, 2011; Northland Regional Council, 2013).

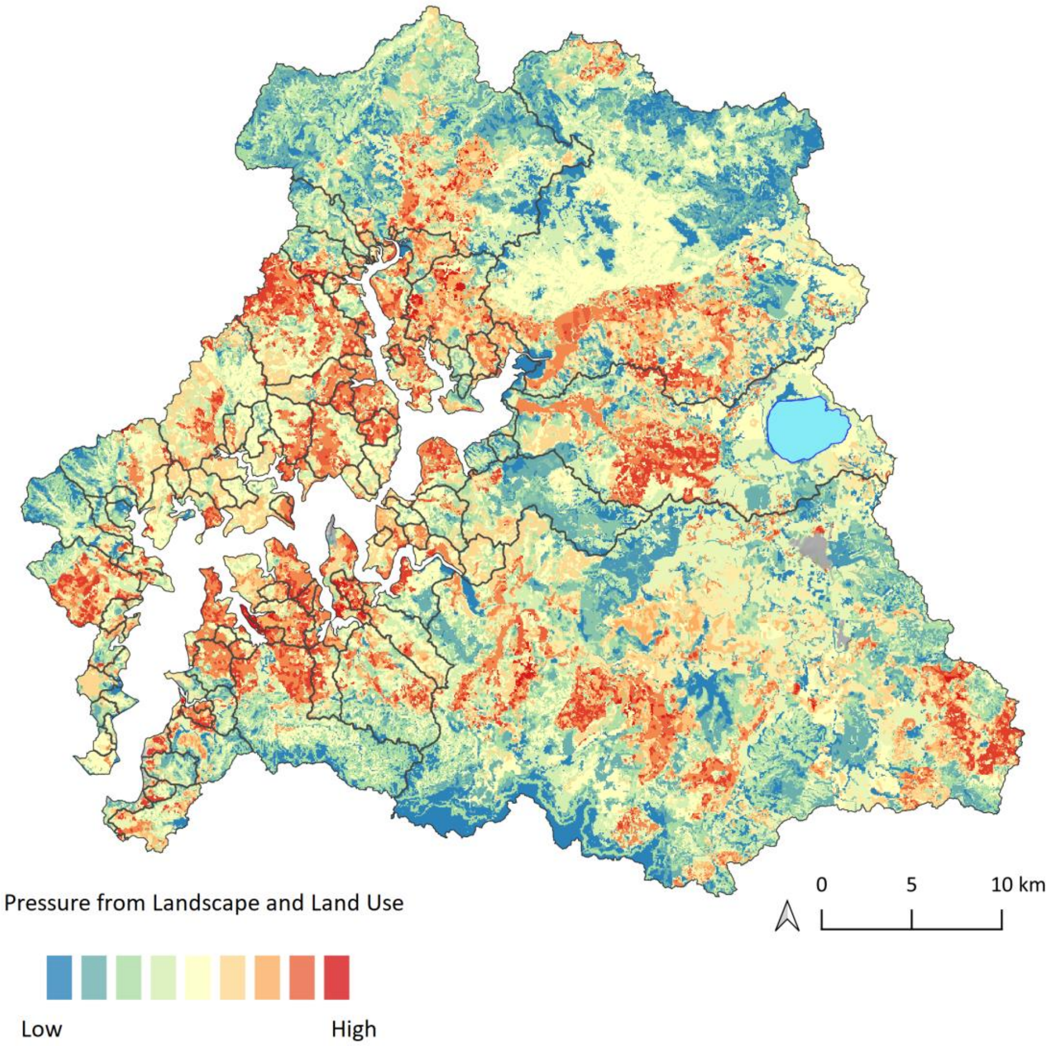


Figure 13. Combined landscape and land use risk to water quality. Where blue areas denote low landscape and land use risk to water quality, and red areas denoted a high combined landscape and land use risk. Mapped from the erosion susceptibility classification of Rissmann et al. (2018b) and the physiographic land intensity layer (ESC + LUI).

In terms of the relationship between land and water, physiographic science indicates that in addition to land use, the landscape plays a critical role in determining the type and severity of water quality across the Hokianga catchment and wider Northland Region. Physiographic modelling suggests that the character of the landscape strongly influences spatial variation in river water quality in addition to land use. Specifically, where land use coincides with the area that is naturally susceptible to erosion (or other) the potential risk to waterways is greatly elevated. Recognising where and why risk occurs is fundamental for effective and least-cost responses to managing water quality and ecosystem health.

In summary, the vital role of the landscape over water quality outcomes has long been recognised, but often poorly characterised. This work advances an overall framework that provides a better understanding of how the landscape influences water quality in addition to land use. Landscape knowledge is vital for guiding investment in mitigations that are appropriately targeted and least cost but also for generating robust policy that is relevant to land users and their communities.

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