Attributes affected by Naturally Occurring Processes in the Bay of Plenty

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By Francis J. Burdon and Deniz Özkundakci

Te Tumu Whakaora Taiao – Environmental Research Institute

Te Whare Wānanga o Waikato – The University of Waikato

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Disclaimer

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Executive Summary

The National Policy Statement for Freshwater Management 2020 (NPS-FM) sets out the objectives and policies for freshwater management under the Resource Management Act 1991 (RMA). The intent of the NPS-FM is to provide local authorities with updated direction on how they should manage freshwater under the RMA, and thus meet their statutory obligations. A fundamental concept in the NPS-FM is *Te Mana o te Wai* which recognises the importance of water to life, and that protecting the health of freshwater enhances the integrity of the wider ecosystem.

The NPS-FM (2020) requires that key attributes for ecosystem and human health are monitored throughout each region. The NPS-FM (2020) is more prescriptive than its 2014 predecessor because it identifies additional freshwaters attributes to manage for. These attributes include water quality, habitat, and biological indicators. Councils must identify baseline and target attribute states (TAS), and if the baseline state is below the National Bottom Line (NBL) for that attribute as set in the NPS-FM, then the target state must be at or above the bottom line. However, the NPS-FM also recognises that naturally occurring processes (NOP) may result in waterbodies not meeting NBLs. These exceptions mean the NPS-FM will allow councils to set TAS worse than a NBL (or Regional Bottom Line), but only if a TAS better than the bottom line is unable to be met due to NOP.

Here we have reviewed evidence from the Bay of Plenty Regional Council (BOPRC) and other sources about three key attributes set forth by the NPS-FM that have been identified as being impacted by NOP in the Bay of Plenty (BOP). These attributes are:

- Dissolved Reactive Phosphorus (DRP; mg/L)
- Lake Bottom Dissolved Oxygen (DO) and Mid-hypolimnetic DO (mg/L)
- Deposited Fine Sediment (DFS; % cover)

In our independent review we have evaluated the potential NOP in the BOP Region and how these NPS-FM attributes (and associated bottom lines) may be inappropriate. We identified NOP for DRP, Lake subsurface DO, and DFS, and assessed what mechanisms underpin these processes, where they are likely to occur in the BOP and their potential impact on attribute states.

In the case of DRP, natural sources of phosphorus associated with geothermal, volcanic and sedimentary geologies are driving high concentrations in BOP streams and rivers. Analysis of national water quality data shows that DRP concentrations are elevated in BOP streams and rivers, despite no differences in areal land cover or a bioindicator of stream health (Macroinvertebrate Community Index). Our results point strongly to high concentrations of DRP being a natural feature of waterbodies in the BOP. With regards to sub-surface lake DO, lake trophic status, morphology, and thermal conditions are likely natural drivers of low DO in the Te Arawa lakes. The naturally high levels of DFS in BOP streams and rivers are largely a consequence of the volcanic geology of the region, with deep layers of tephra, ash and fine pumice leading to streams dominated by fine substrates. We present results from a survey of streams in the central North Island, where naturally high levels of DFS decoupled this attribute with macroinvertebrate bioindicators of ecosystem health.

Our report has covered these attributes where the stated NOPs mean that the baseline states are below NBL, thus providing supporting evidence in accordance with clause 3.32(2) in the NPS-FM. Our advice helps to begin developing realistic TAS for these attributes, meaning that the BOPRC can seek improvements in environmental health that are underpinned by the best available knowledge and reflect the unique natural features of the Bay of Plenty.

Contents

Background and purpose

Regional councils have statutory obligations under the Resource Management Act 1991 (RMA) and the more recent National Policy Statement on Freshwater Management 2020 (NPS-FM) to assess the ecological state of waterways and monitor trends over time. Under Section 35 of the Resource Management Act, Regional Councils have "a duty to monitor the state of the whole or any part of the environment to the extent that is appropriate to enable the local authority to gather information, monitor and keep records of any necessary information required to effectively carry out their functions under the act."

More recently under the NPS-FM (2020), the National Objectives Framework (NOF) has been created so that councils can:

- set environmental outcomes for identified values and include them as objectives in regional plans
- identify attributes for each value and identify baseline states for those attributes
- set target attribute states to support the achievement of environmental outcomes
- set limits as rules and prepare action plans (as appropriate) to achieve environmental outcomes

The NPS-FM also sets forth specific responsibilities where Regional Councils must monitor attributes identified in Appendices 2A and 2B. These include 10 attributes that require limits on resource use and 12 attributes that require action plans to achieve environmental outcomes. The latter must also meet target attribute states that at minimum maintain the ecological state of the system but also reflect better conditions than that described by National Bottom Lines (NBLs). Specifically, Clause 3.11(2) of the NPS-FM states that the target attribute state (TAS) for every value with attributes need to set at or better than the baseline state of that attribute. These attributes are designed so that if a particular attribute state is worse than a TAS – set as objectives – then councils must set rules to help limit resource use to achieve the TAS (Appendix 2A attributes) or prepare an action plan for achieving the TAS (Appendix 2B attributes). Communities and tangata whenua must be consulted and engaged with at each step of the NOF process.

However, the NPS-FM (Clause 3.32) recognises that naturally occurring processes may result in waterbodies not meeting NBLs. These exceptions mean the NPS-FM will allow councils to set TAS worse than a NBL (or Regional Bottom Line) only if a TAS better than the bottom line is unable to be met due to naturally occurring processes. Determining what a naturally occurring process is and how it affects attributes requires the best available information. Scientists from the Bay of Plenty Regional Council (BOPRC) have reviewed the attributes set forth by the NPS-FM and identified naturally occurring processes in the Bay of Plenty that impact several of these. These natural processes in the Bay of Plenty may affect the capacity for these attributes to meet bottom lines. Reviews by BOPRC scientists have documented such issues for:

- Dissolved Reactive Phosphorus (DRP) in Dare (2019) and Scholes (2021);
- Lake Bottom Dissolved Oxygen (DO) and Mid-hypolimnetic DO in Scholes (2021); and
- Deposited Fine Sediment (DFS) in Suren (2020)

In addition, parts of the Bay of Plenty are subjected to geothermal activity that may also impact TAS for NPS-FM attributes.

Consequently, BOPRC is seeking an independent review of where naturally occurring processes in the Bay of Plenty Region mean the applicability of these NPS-FM attributes (and associated bottom lines) are inappropriate and beyond the control of the council. The review we have undertaken specifically addresses the naturally occurring processes identified in the publications mentioned above (Dare 2019, Scholes 2021, Suren 2020). It also considered any other information relevant to successfully implementing the NPS-FM in the Bay of Plenty. The report covers the attributes where naturally occurring processes mean that the current and baseline states are below bottom lines, and provides supporting evidence in accordance with clause 3.32(2) in the NPS-FM.

For attributes where baseline states do not meet bottom lines due to natural causes, we have attempted to estimate the proportion of the baseline state that is likely caused by naturally occurring processes (where the state is a concentration), and/or the likely naturally occurring state. We have provided guidance on the development of alternative attribute state tables. The rationale for these recommendations regarding alternative attribute state bands are discussed in the report.

Dissolved Reactive Phosphorus

Context

Phosphorus occurs in freshwater as different forms – orthophosphate dissolved in water and attached to inorganic particles, as dissolved organic molecules and in particulate organic form (Allan and Castillo 2007). Where total phosphorus (TP) is analysed using unfiltered water samples and represents all forms of phosphorus (P), dissolved reactive phosphorus (DRP) is measured from filtered samples using the reaction of soluble P with the compound molybdate. DRP is generally regarded as the best indicator of what P is immediately available for uptake by organisms, although when P cycles rapidly between its various states TP may be a better measure of overall P availability.

In contrast to the atmospheric sources of nitrogen, the main natural sources of P include rocks and sediment. Thus, catchment geology and rock weathering are important drivers of this nutrient in freshwater ecosystems. Some sedimentary and volcanic rocks can be P rich, whereas crystalline rocks (e.g., intrusive igneous rocks) are typically P deficient leading to variation in the DRP concentrations of streams draining catchments with different geologies (Allan and Castillo 2007). Streams influenced by geothermal activity can also have elevated concentrations of DRP. For example, Beyá et al. (2005) recorded an average DRP concentration of 0.290 mg/L from a geothermal stream in the Lake Rotoiti catchment – a concentration much higher the global average of 0.008 mg/L (Meybeck 1982).

Phosphorus is a common constituent in agricultural fertilizers, manure, and the organic wastes in sewage and industrial effluent, meaning diffuse and point-sources of pollution associated with human activities often lead to increased concentrations in freshwaters and receiving habitats. Consequently, DRP is often in the range of 0.05 — 0.1 mg/L in streams receiving agricultural runoff but can reach 1 mg/L below sewage outfalls (Allan and Castillo 2007). In New Zealand, differences in geology and the extent of agricultural activities mean average concentrations of DRP can vary widely (e.g., 0.009 — 0.074 mg/L) by region (McDowell et al. 2009).

Whilst phosphorus (P) is an essential nutrient for all plant growth, excessive concentrations of DRP can lead to eutrophication of waterbodies typified by nuisance growths of periphyton, macrophytes, and/or phytoplankton. Such unwanted primary production in aquatic systems negatively affects ecosystem health and services (e.g., amenity values that enhance people's use and enjoyment of the water body). Key ecosystem functions can be altered including how microbes and invertebrates break down and recycle organic matter (such as leaf litter) in rivers (Burdon et al. 2020). Altered decomposition processes can lead to increased respiration and decreased dissolved oxygen concentrations.

Excessive P can affect all freshwater ecosystems including rivers, wetlands, lakes and estuaries. However, these impacts may be most relevant in larger rivers, lakes, and other waterbodies with higher retention rates, infrequent disturbance, and abundant photosynthetically available radiation (light). In contrast, shaded and hydrodynamically disturbed (i.e., bed-moving floods) streams may be more resilient to inputs of phosphorus due to other factors (light, disturbance) limiting primary production (Biggs 2000). Factors that can exacerbate the impacts of bed movement in floods include the grain size of the benthic substrate, with fine sediment more prone to entrainment and scouring (Jones et al. 2012). However, when streams have stable flows relatively unaffected by bed-moving floods, rooted macrophytes can get their P requirements from deposited sediment (Sand-Jensen 1998). This suggests that sedimentation may exacerbate nuisance growths of these plants in situations where other factors (e.g., light) are not limiting, further decoupling the relationship between DRP and unwanted plant growth.

Table 20 – Dissolved reactive phosphorus

Value (and component)	Ecosystem health (Water quality)	
Freshwater body type	Rivers	
Attribute unit	DRP mg/L (milligrams per litre)	
Attribute band and description	**Numeric attribute state**	
	Median	95th percentile
A Ecological communities and ecosystem processes are similar to those of natural reference conditions. No adverse effects attributable to dissolved reactive phosphorus (DRP) enrichment are expected.	≤ 0.006	≤ 0.021
B Ecological communities are slightly impacted by minor DRP elevation above natural reference conditions. If other conditions also favour eutrophication, sensitive ecosystems may experience additional algal and plant growth, loss of sensitive macroinvertebrate taxa, and higher respiration and decay rates.	> 0.006 and ≤0.010	> 0.021 and ≤0.030
C Ecological communities are impacted by moderate DRP elevation above natural reference conditions. If other conditions also favour eutrophication, DRP enrichment may cause increased algal and plant growth, loss of sensitive macro-invertebrate and fish taxa, and high rates of respiration and decay.	> 0.010 and ≤ 0.018	> 0.030 and ≤ 0.054
D Ecological communities impacted by substantial DRP elevation above natural reference conditions. In combination with other conditions favouring eutrophication, DRP enrichment drives excessive primary production and significant changes in macroinvertebrate and fish communities, as taxa sensitive to hypoxia are lost.	>0.018	>0.054
Based on a monthly monitoring regime where sites are visited on a regular basis regardless of weather and flow conditions. Record length for grading a site based on 5 years.		

Figure 1 Table 20 from Appendix 2B of the National Policy Statement for Freshwater Management (NPS-FM) 2020 outlining the attribute bands for Dissolved Reactive Phosphorus (DRP).

Dissolved Reactive Phosphorus (DRP) attribute in the NPS-FM

Phosphorus is typically measured as total phosphorus (TP) and dissolved reactive phosphorus (DRP). Most phosphorus in waterways is bound to sediment and not readily available for plant growth. The dissolved fraction is highly labile and easily assimilated by plants. Consequently, DRP is an important attribute in the NPS-FM 2020 requiring the BOPRC to measure and monitor its concentrations (see Table 20 in the NPS-FM but provided as Figure 1 above). Nutrient outcomes need to achieve target attribute states meaning the instream concentrations and exceedance criteria, or instream loads, for phosphorus. Councils are required to manage concentrations in rivers to provide for other attributes such as periphyton. Periphyton usually grows in hard-bottomed (stony or gravelly) rivers and streams but may also potentially foul soft-bottomed (i.e., pumice gravels and sand, silt and mud) waterways when the bed-moving flood frequency is insufficient to prevent biomass accrual (Biggs 2000). Standard methods to assess periphyton biomass have only been developed for hard-bottomed streams however (NEMS 2022), and it is presumably only hard bottomed streams that are the focus of the

NPS-FM. Councils are also required to manage for objectives in nutrient-sensitive downstream ecosystems such as lakes and estuaries. DRP is measured as concentrations (mg/L). The attribute bands in the NPS-FM rely on median and 95th percentile values that are based on a monthly monitoring regime. Sites are visited on a regular basis regardless of weather and flow conditions, and the record length for grading a site is based on five years of monthly monitoring. BOPRC currently monitors DRP monthly across 51 stream and river sites in the Bay of Plenty as part of the Natural Environment Regional Monitoring Network (NERMN) programme (Scholes 2021).

In Clause 3.13 of the NPS-FM there are special provisions for attributes affected by nutrients. The first of these holds that: "To achieve a target attribute state for any nutrient attribute, and any attribute affected by nutrients, every regional council must, at a minimum, set appropriate instream concentrations and exceedance criteria, or instream loads, for nitrogen and phosphorus." However, periphyton communities in the BOP region appear to be only weakly controlled by the nutrient Total Nitrogen (TN), with the mean annual frequency of floods > 2 (FRE2), temperature and substrate size being more important influence factors (Kilroy et al. 2020). Of these predictor variables, TN was the weakest, and substrate size and temperature the strongest. Furthermore, they found that when DRP was included as a predictor variable in their models, model performance declined. These results suggest that the effects of nutrients on periphyton in the BOP region are weak, particularly for DRP. Kilroy et al. (2020) also contended that periphyton in the region was not DRP limited, as median DRP concentrations exceeded the literature values of saturating concentrations, thus supporting the contention that the other influencing factors drive periphyton accrual.

The second provision in Clause 3.13 of the NPS-FM holds that where there are nutrient-sensitive downstream receiving environments, the instream concentrations and exceedance criteria, or the instream loads, for nitrogen and phosphorus for the upstream contributing water bodies must be set to achieve the environmental outcomes sought for the downstream environments. This clause is of particular importance for the Bay of Plenty with potentially nutrient-sensitive downstream receiving environments including the Te Arawa Lakes and nationally important estuaries such as the Tauranga, Kaituna-Maketu and Ōhiwa Harbours. The third provision holds that in "setting instream concentrations and exceedance criteria, or instream loads, for nitrogen and phosphorus under this clause, the regional council must determine the most appropriate form(s) of nitrogen and phosphorus to be managed for the receiving environment."

However, although the third provision provides some discretionary powers, the fourth provision states that "every regional council must adopt the instream concentrations and exceedance criteria, or instream loads, set under subclauses (1) and (2) as nutrient outcomes needed to achieve target attribute states." This latter provision is of particular importance, since excessive nutrients can affect other attributes under the NPS-FM highlighted by the fifth provision including periphyton, dissolved oxygen (see NPS-FM 2020, Appendix 2A, Tables 2 and 7 and Appendix 2B, Tables 17, 18, and 19), submerged plants (invasive species) (Appendix 2B, Table 12), fish (rivers) (Appendix 2B, Table 13), macroinvertebrates (Appendix 2B, Tables 14 and 15), and ecosystem metabolism (Appendix 2B, Table 21). The central importance of nutrients in the NPS-FM as a "master variable" means that assessing the relevance of this attribute (i.e., DRP) in the BOP region is a key function of this review. However, the best information available suggests that nutrients are not the most important driver of freshwater ecosystem health in the BOP region (Snelder et al. 2019, Kilroy et al. 2020). This present understanding does not diminish the ongoing importance of managing nutrient inputs from diffuse and point sources for the BOPRC, given the potential for multiple stressor interactions to cause undesirable ecological "surprises" (Matthaei et al. 2010, Jackson et al. 2016) and impacts on sensitive downstream receiving envrionments.

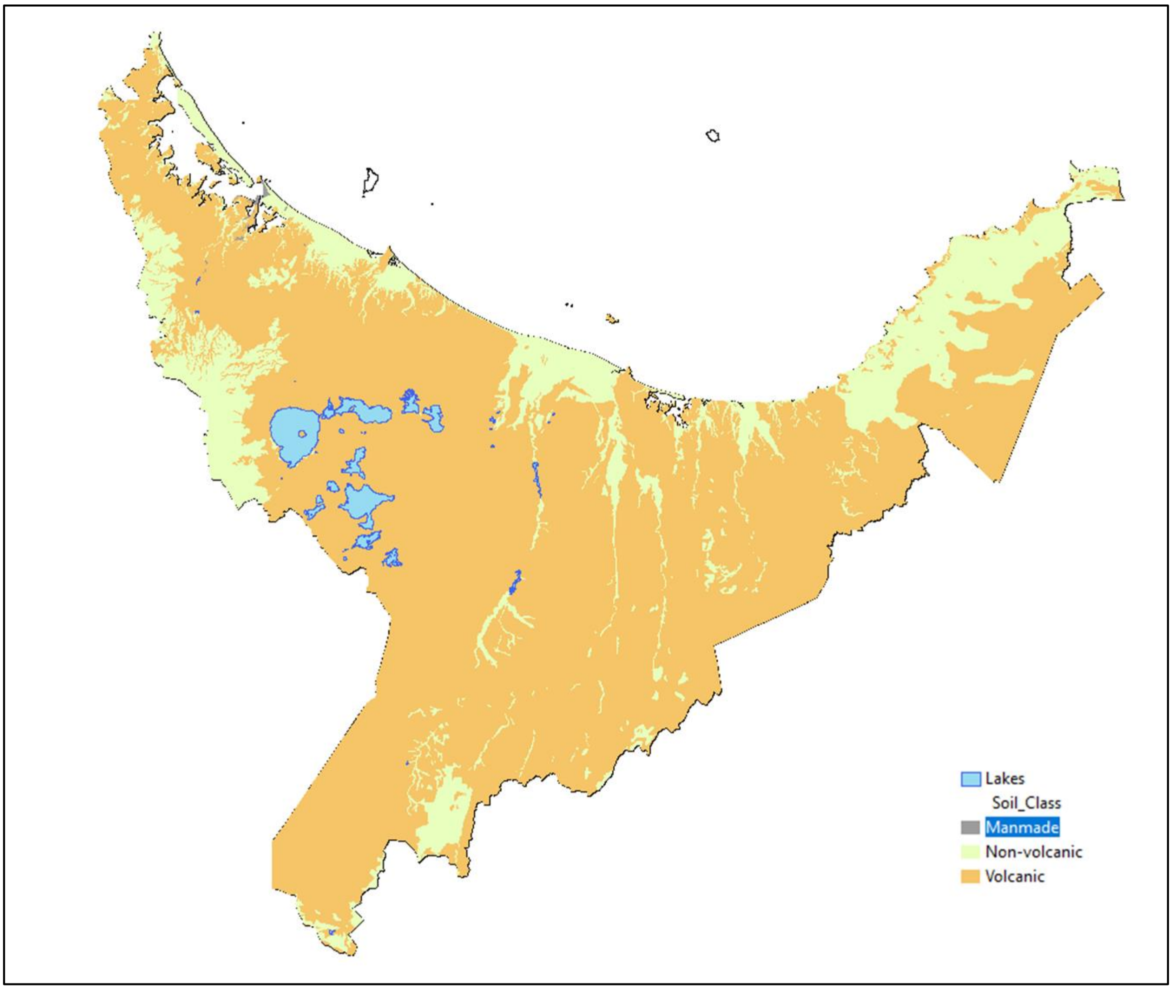

Figure 2. Map of the Bay of Plenty region showing soil type overwhelmingly dominated by soils of volcanic origin (orange). These include the layers of tephra (e.g., pumice and ash) from past volcanic activity that are characteristic of region. This map was generated by Alastair Suren (BOPRC) using the council's soil layer in ArcGIS software.

Current status of DRP in the BOP

Dare (2019) found that over 50% of the monitoring sites in the BOP region performed poorly for the DRP attribute (i.e., in attribute band "D"). The poor status implied by the DRP attribute in the BOP is strongly influenced by naturally occurring processes. In the central BOP, the enrichment of groundwater through rock weathering associated with volcanic geologies alongside surface and subterranean geothermal inputs of P may help explain elevated levels of DRP in streams of this subregion (Table 1). The soil layer in the BOPRC's Geographic Information System (GIS) software (ArcGIS) indicates the widespread distribution of volcanically derived, pumice-dominated soils (Figure 2). This distribution notably includes parts of the eastern BOP, where the underlying parent geology is non-volcanic. The widespread tephra layer that includes fine pumice is likely from historic volcanic eruptions (e.g., the Kaharoa eruption from the Tarawera volcano within Haroharo caldera, the Hatepe eruption in the Taupō Volcano, and the eruption of the Rotorua caldera), with this material extensively deposited over volcanic and non-volcanic basement rocks in the region (Lowe 1990). High phosphate retention in allophanic soils formed from layers of volcanic ash in the Rotorua and eastern BOP areas may mitigate some potential for P leaching (Rijkse and Guinto 2010). Instead, the gradual dissolution of DRP to groundwater in the porous pumice layers (Tempero et al. 2015) and the erodibility of the volcanic soils exporting sediment-transported P (McDowell 2010) help to explain elevated DRP concentrations in receiving stream and rivers. In the eastern BOP, sedimentary sources of phosphates might also explain some of elevated concentrations of DRP observed in predominately forested catchments (Table 1). Climatic effects include rainfall and subsequent erosion of volcanic and sedimentary geologies potentially also influences DRP concentrations (Table 1). In some exceptional circumstances, high abundances of feral mammals may also contribute to increased DRP via decreased vegetation cover, greater sediment erosion, and subsequent regeneration in stream sediments (Table 1). The different naturally occurring processes summarised in Table 1 potentially give the erroneous appearance that streams in the BOP region are characterised by inputs of P derived from anthropogenic activities leading to poor status under the NPS-FM. In the following section we have provided an overview of the DRP situation in the BOP, before summarising the BOPRC assessments regarding DRP to date with reference to the wider literature. We have thoroughly considered the naturally occurring processes potentially leading to elevated DRP concentrations in the region with concluding remarks on the potential way forward.

The hierarchical River Environment Classification (REC) is an important tool for riverine habitat classification in New Zealand (Snelder and Biggs 2002). The REC describes a range of natural factors influencing water quality (e.g., land cover, climate, topography and geology). It is widely used for understanding water quality patterns in New Zealand (Larned et al. 2005) but is not without limitations (Snelder et al. 2004). In combination with REC classifications, we analysed publicly available data from the LAWA website (www.lawa.org.nz) to show that average DRP concentrations are higher in Bay of Plenty streams and rivers draining catchments dominated by native and exotic forest land cover (Figure 3, Table 2). DRP concentrations in native forest catchments did not differ significantly from their equivalents in the rest of the North Island but were significantly higher than those in the South Island ($p<0.01$). DRP concentrations recorded from exotic forest catchments in the BOP were significantly higher than those in the rest of North Island ($p<0.05$) and the South Island ($p<0.01$). In contrast, DRP concentrations in BOP rivers draining pastoral catchments were no different from those in the other parts of New Zealand. A striking result in the BOP was that DRP concentrations in native forest catchments did not differ from pastoral catchments ($p=0.91$), in contrast to clear differences between these two land cover types in the rest of North Island ($p<0.001$) and the South Island ($p<0.001$). Another exceptional result in the BOP was that exotic forest catchments had significantly higher DRP concentrations than native forest *and* pastoral catchments (both $p<0.05$). DRP concentrations from exotic forest catchments in the other parts of New Zealand did not differ from other land cover types.

All monitored exotic forest catchments in the Bay of Plenty ($n=7$) are characterised by volcanic geology in the REC database (Table 3), and all but one are in the Cold-Wet climate group. When comparing the BOP with other parts of New Zealand, the areal proportion (%) of catchment land cover by different REC categories did not differ (Table B1). Similarly, Macroinvertebrate Community Index (MCI) scores did not differ between the Bay of Plenty and the other parts of New Zealand (Table B2). The lack of congruence between land cover, macroinvertebrate responses and the elevated DRP concentrations observed in the BOP are strong evidence for naturally occurring processes contributing to the surfeit of this nutrient in rivers and streams of the BOP.

Table 1. Naturally occurring processes (NOP) that can cause elevated concentrations of dissolved reactive phosphorus (DRP) in stream and rivers. The areas that are likely affected by NOP in the Bay of Plenty (BOP) region are indicated, including River Environment Classification (REC) landcover and underlying geology. The Taupō Volcanic Zone (TVZ) is volcanically active and runs north-eastward from Mount Ruapehu through the Taupō and Rotorua areas and offshore into the Bay of Plenty. Feral mammals in the BOP include wild populations of Dama wallaby (*Macropus eugenii*), along with red deer (*Cervus elaphus*), fallow deer (*Dama dama*), pigs (*Sus scrofa*) and goats (*Capra hircus*). Regeneration of dissolved P can be enhanced through anoxic conditions in fine riverine sediments (Mainstone and Parr 2002). Arrows indicate direction of change from attribute A status (i.e., going up to B-D status).

Naturally Occurring Process		Mechanism	Key references	Where in the BOP (REC landcover, underlying geology)	Likely occurrence/ importance in the BOP	Likely effect on DRP attribute band (relative to A status)
Geological	Geothermal	Disassociation of P into groundwater through rock weathering and geochemical processes. Transport in geothermal fluids from source to springs, streams, and rivers	Beyá et al. (2005), Pringle et al. (1993), Triska et al. (2006), Hoellein et al. (2012)	Pasture and Forest (Exotic, Native) within the Taupō Volcanic Zone (TVZ)	Medium-High	↑ C-D
	Volcanic	Disassociation of P into groundwater through rock weathering and geochemical processes, also erosion and transport via runoff to streams	Morgernstern et al. (2015), Munn and Meyer (1990), McDowell et al. (2004)	Pasture and Forest (Exotic, Native) with underlying volcanic bedrock and/or soils	High	↑ B-D
	Sedimentary	Disassociation of P through rock weathering and erosion, regeneration of dissolved P via biogeochemical processes in riverine sediments	Dillon & Kircher (1975), Porder and Ramachandran (2012), Thomas and Crutchfield (1974)	Pasture and Forest (Exotic, Native) with underlying soft-hard sedimentary bedrock (mudstones/ limestones)	Medium	↑ B-D
Climate		Increased erosion associated with more variable rainfall, subsequent regeneration of dissolved P via biogeochemical processes in deposited riverine sediments	Ramos et al. (2022), Scarsbrook et al. (2003)	All areas with wet climates, particularly those erosion-prone associated with sedimentary bedrock	Medium-Low	↑ B
Biological	Feral Mammals	Reduced vegetation cover and increased erosion, subsequent regeneration of dissolved P via biogeochemical processes in deposited riverine sediments	Scanes et al. (2021), Wills et al. (2023)	Forest (Exotic, Native) with volcanic and/or sedimentary soils	Low	↑ B

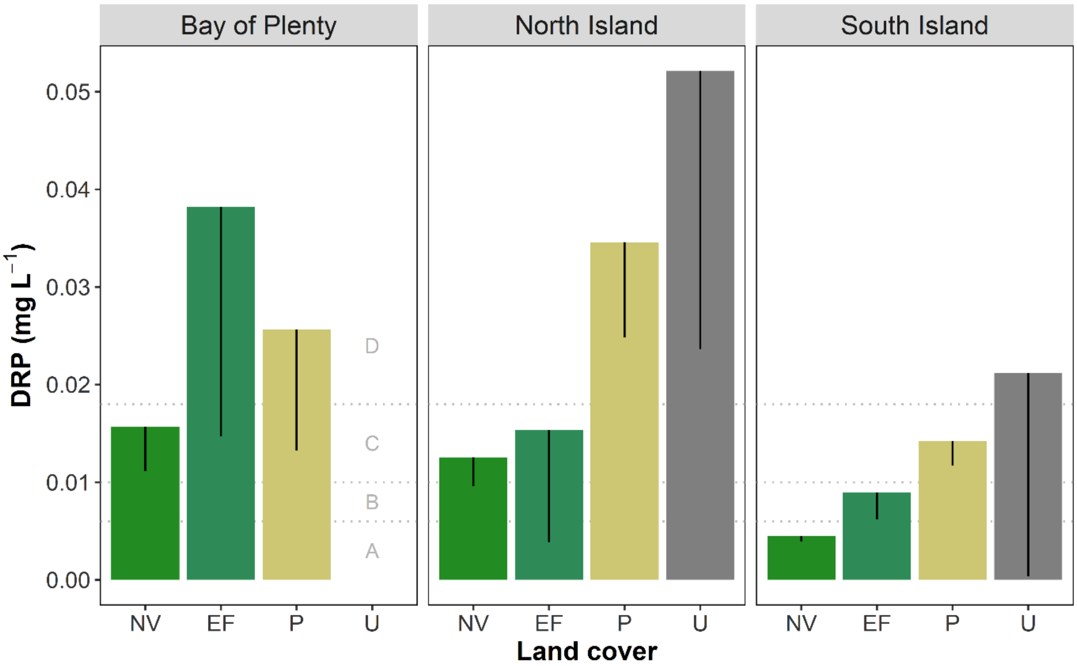

Figure 3. Mean concentrations of dissolved reactive phosphorus (mg/L) in streams and rivers in New Zealand. Sites have been grouped into North and South Islands to compare with the Bay of Plenty Region. The North Island sites excluding the Bay of Plenty include the Auckland, Gisborne, Hawkes Bay, Wellington, Manawatū-Whanganui, Northland, Taranaki, and Waikato regions. Land cover indicates broad categories assigned in the River Environment Classification (NV, Native vegetation; EF, Exotic forestry; P, Pasture; U, Urban). Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz). The black bars indicate the lower range of the 95% confidence intervals; for the upper range see Table 2.

We used LAWA data for DRP concentrations in BOP rivers over the past five years to consider the REC climate and source of flow categories in addition to land cover and geology. Using a linear mixed-model approach with a random effects term for sampling location nested in catchment, there were significant differences among land cover, source of flow, and climate but surprisingly not geology. Native vegetation sites had lower concentrations of DRP than exotic forest sites ($p<0.05$). Climate had a significant effect on DRP concentrations in the BOP, with the warm-wet climate sites having higher concentrations than the warm-extremely wet and the cold-wet climate groups ($p<0.001$, Figure B1). This confirmed a basic trend observed by Larned et al. (2005) of water quality increasing with colder and higher rainfall climates. Hill source of flow streams had lower DRP concentrations than lowland or lake outlets ($p<0.05$). Using the "Topography" group (source of flow and climate combined) following McDowell et al. (2013) revealed a strong influence of these drivers on DRP concentrations in the BOP ($p<0.001$), with the land cover influence disappearing. This suggests that the source of flow (three exotic forest sites are lake outlets) and to a lesser extent climate of exotic forest sites in the BOP explain their higher concentrations of DRP relative to the native vegetation sites in this region.

Overall, our results using the LAWA data suggest that many of the elevated DRP concentrations recorded in the Bay of Plenty are the result of naturally occurring processes, since land cover does not differ significantly from other regions of New Zealand, and macroinvertebrate indicators did not show a concomitant loss in stream ecosystem health.

Table 2. Mean and median concentrations (± 95% confidence intervals) of dissolved reactive phosphorus (mg/L) in streams and rivers in New Zealand. Sites have been grouped into North and South Islands to compare with the Bay of Plenty Region. The North Island sites excluding the Bay of Plenty include the Auckland, Gisborne, Hawkes Bay, Wellington, Manawatū-Whanganui, Northland, Taranaki, and Waikato regions. Land cover indicates broad categories assigned in the River Environment Classification database. Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz).

Group	Land cover	Sites	Mean	Lower CI	Upper CI	Median	Lower CI	Upper CI
Bay of Plenty	Native veg.	16	0.016	0.011	0.020	0.017	0.011	0.023
	Exotic forest	7	0.038	0.015	0.062	0.025	-0.018	0.033
	Pasture	28	0.026	0.013	0.038	0.013	0.005	0.017
	Urban	0						
North Island	Native veg.	91	0.013	0.010	0.016	0.008	0.005	0.010
	Exotic forest	12	0.015	0.004	0.027	0.007	-0.007	0.009
	Pasture	409	0.035	0.025	0.044	0.014	0.012	0.015
	Urban	28	0.052	0.024	0.081	0.022	0.009	0.030
South Island	Native veg.	138	0.005	0.004	0.005	0.004	0.004	0.004
	Exotic forest	10	0.009	0.006	0.012	0.007	0.002	0.008
	Pasture	238	0.014	0.012	0.017	0.008	0.007	0.010
	Urban	13	0.021	0.000	0.042	0.011	0.006	0.012

Dare (2019) used data from 45 of the NERMN sites (extracted for the five-year period spanning 1st April 2014 to 1st April 2019) to generate summary statistics for the relevant attributes. This data analysis included a qualitative assessment of whether each NERMN site might be influenced by naturally high concentrations of DRP. This assessment was based on substrate type and geology, where sites situated in volcanic soft sediment catchments are most likely to have elevated natural DRP concentrations. Eighteen sites (40%) were assessed to be 'Natural DRP' (influenced by naturally high levels of volcanic derived DRP) based on the qualitative assessment by Dare (2019). This number of sites (18) is a subset of the 23 sites (≈51%) deemed to be in the "D" band for the DRP attribute, which based on national guidelines are indicative of ecological communities impacted by substantial DRP elevation above natural reference conditions. DRP enrichment may drive excessive primary production leading to eutrophication when other factors are not limiting. Eutrophication can lead to significant changes in macroinvertebrate and fish communities, as taxa sensitive to hypoxia are lost, although this outcome seems less relevant in the BOP region due to other factors limiting periphyton biomass accrual and macroinvertebrate diversity (Snelder et al. 2019, Kilroy et al. 2020). Scholes (2021) noted that many of the sites determined to be influenced by naturally high concentrations of DRP are at or near the bottom of catchments and are potentially subject to a range of anthropogenic impacts. However, Scholes (2021) also recognises that recent surveys of springs across the BOP region, many of which are near headwaters of streams, are consistent with Dare's (2019) observations. The differences in DRP concentrations between springs and river sources are shown in Figure C1. For the spring sites, 65% of the 83 sites surveyed were in the "D" band, indicating the likelihood of natural sources of DRP.

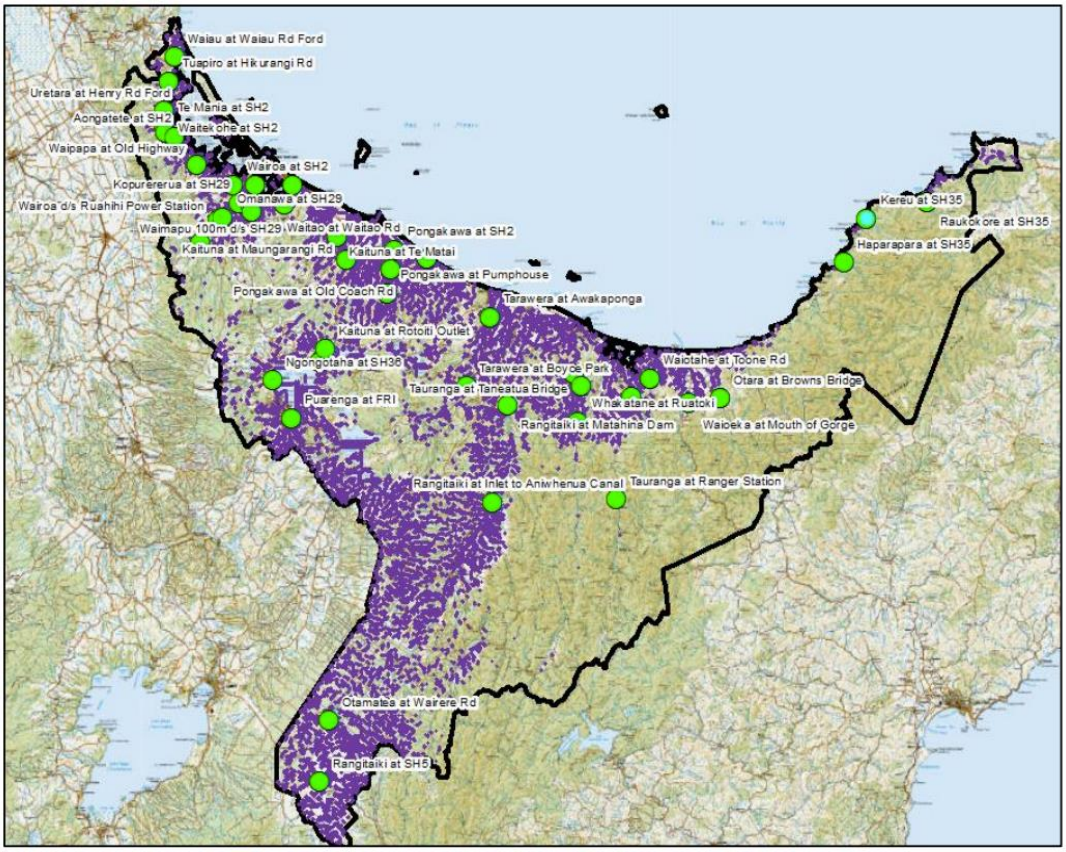

Figure 4. Map of the Bay of Plenty region from Dare (2019) showing Natural Environment Regional Monitoring Network (NERMN) water quality sites in green and volcanic/soft bottom geology highlighted in purple. Sites present in purple areas are likely to have elevated concentrations of natural DRP due to NOP.

Dare (2019) also provided a map of stream reaches that are in the 'soft-volcanic' substrate/geology category according to the Freshwater Ecosystems New Zealand (FWENZ) spatial model overlaid with the NERMN sites in the BOP. This map, reproduced in Figure 4 shows the sites (in the purple reaches) that are expected to have naturally high DRP concentrations due to the underlying volcanic geology. Sites near their headwaters with spring sources are more likely to be affected by enrichment of underlying groundwater through porous volcanic geology (Scholes 2021). However, this process is highly dependent on the geological origins of the contributing groundwater sources and the contact time with relevant geological formations. Thus, determining the contribution of groundwater to surface flows also must consider the residence times of subterranean flows and the biogeochemical attributes determining the concentration of emerging soluble phosphorus. The relationship between mean residence time and DRP concentrations described by Morgenstern et al. (2015) in the Rotorua catchment was highlighted by Scholes (2021). The log-linear nature of the relationship can be demonstrated using the following two examples: in the period 2016-2020 Hamurana Springs had a mean residence time (MRT) of 145 years and DRP of around 0.1 mg/L, whereas the Ngongotahā Stream had a MRT of 30 years and DRP median concentrations of 0.026 mg/L (Scholes 2021).

Natural geothermal inputs may also cause elevated concentrations of DRP in receiving streams. Streams in the Te Arawa Lakes region can have geothermal inputs (e.g., the Puarenga Stream) that demonstrate naturally occurring processes leading to high DRP. For example, the Ōtuatura Stream is a geothermal spring-fed tributary of Lake Rotoiti that has a reported average DRP concentration of 0.290 mg/L (Beyá et al. 2005). Monthly physicochemical data for the surface water tributaries to Lakes

Rotorua and Rotoiti between 1990 and 2003 showed a distinct elevation of DRP and TP (Hoellein et al. 2012). Those authors showed that geothermal streams in the Te Arawa Lakes region had concentrations of DRP that were over 2.5 times higher than non-geothermal streams (Figure C2). Similar studies overseas confirm these trends. A study by Pringle et al. (1993) in a volcanic region of Costa Rica showed that spring inputs associated with geothermal activity led to greatly increased mean concentrations of DRP (e.g., 0.056—0.088 mg/L). These elevated concentrations could be contrasted with background concentrations (0.08 µg/L) in streams draining catchments with highly weathered lavas and covered in tropical rainforest (Pringle et al. 1993, Triska et al. 2006).

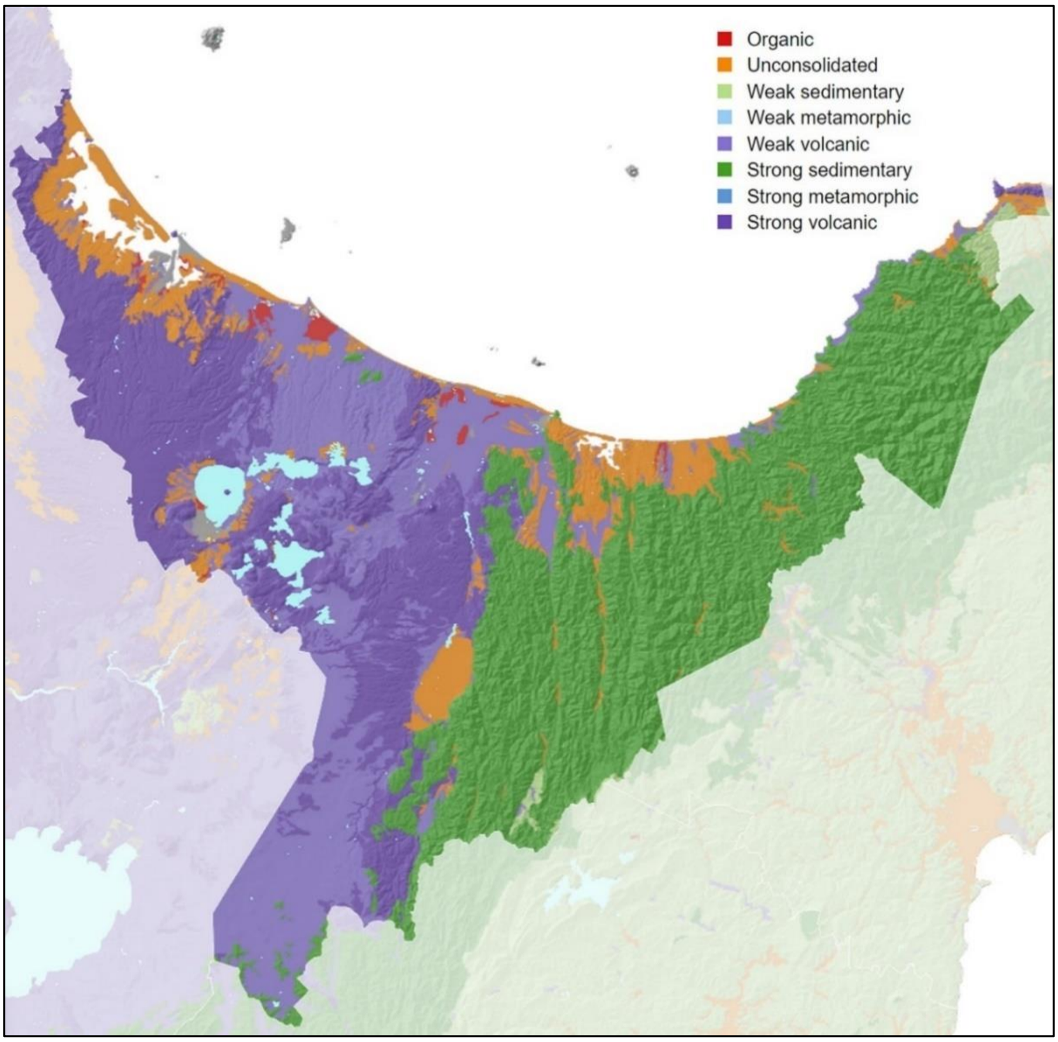

Figure 5. Map of the Bay of Plenty region showing Base Rock Strength groups according to Newsome (1992). The green areas indicated where sedimentary rocks (e.g. greywacke) dominate the underlying geology of the region. This information was generated from the www.landscapedna.org portal.

Streams draining hard sedimentary geologies in the eastern BOP (Figure 5) may be influenced by naturally occurring sources of sedimentary P (Scholes 2021), and/or reflect layers of volcanic material ejected from past eruptions (e.g., Figure 2). BOP streams and rivers in hard sedimentary catchment geologies have a median DRP concentration of 0.18 mg/L (0.012−0.024 mg/L 95% CI), despite the dominant land cover being native forest (Figure 6). This is well within the range of the D band (>0.018

mg/L). Catchments such as those in the eastern BOP are typified by basement greywacke deposits, a sedimentary rock common to mountainous regions of New Zealand. Subsurface flows in forested, greywacke-dominated catchments of New Zealand typically have low concentrations of DRP (Dymond et al. 2013), although White (1972) found evidence for elevated concentrations of DRP in a North Island catchment which he attributed to a potential sedimentary rock source in a faulted section of greywacke. A study in the southern Canadian Shield region of North America reported that the P export from streams draining watersheds of sedimentary origin exceeded that of igneous and metamorphic sources (Dillon and Kirchner 1975). Sedimentary catchments with phosphate-bearing limestone can contain substantially more P in streams when compared to streams draining catchments where sandstones and shales dominate the geology (Thomas and Crutchfield 1974).

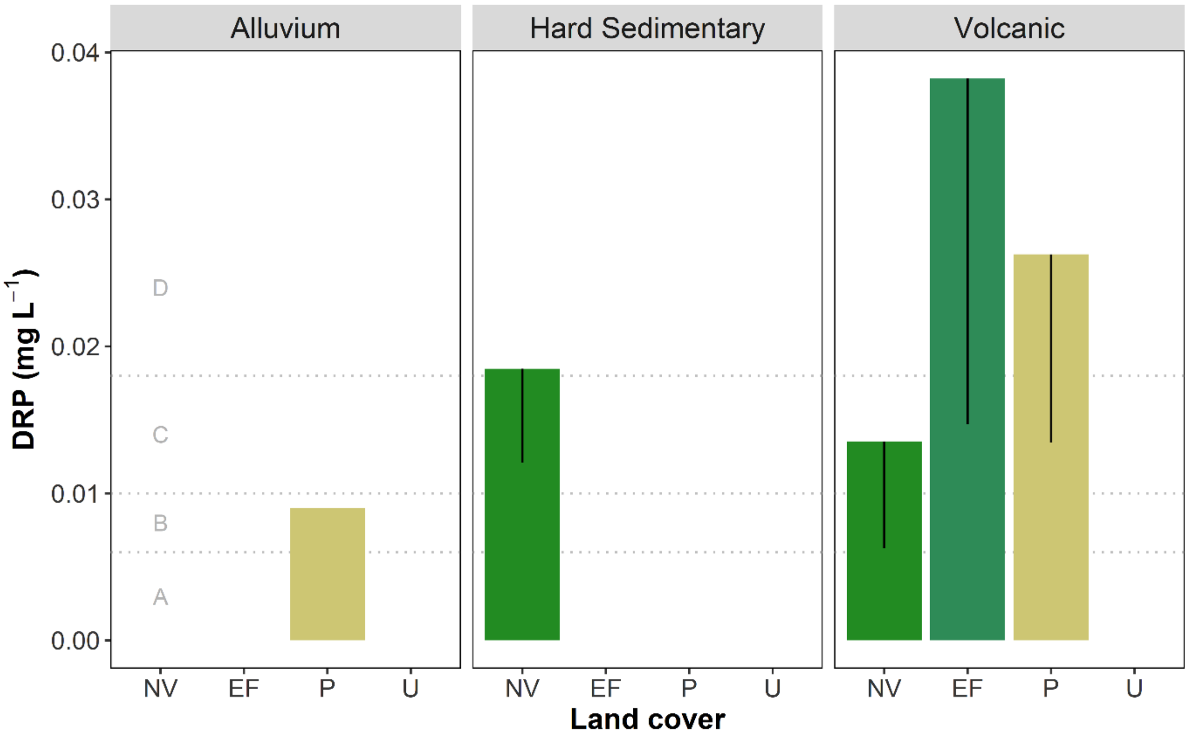

Figure 6. Mean concentrations of dissolved reactive phosphorus (mg/L) in Bay of Plenty streams and rivers. Sites have been grouped into categories based on catchment geology described in the River Environment Classification (REC) database. Land cover indicates broad REC categories (NV, Native vegetation; EF, Exotic forestry; P, Pasture; U, Urban). Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz), REC classifications for catchment geology from the MfE data portal (www.data.mfe.govt.nz). The black bars indicate the lower range of the 95% confidence intervals; for the upper range see Table 3.

There are limestone deposits in the eastern BOP that could be sedimentary sources of DRP. For instance, limestone was previously quarried near Ruatoki south of Whakatāne, and deposits can also be found southeast of Murupara (Healey 2005). The limestone deposits in the eastern BOP appear to be Cambrian to Cretaceous basement terranes (Mortimer and Strong 2014). A historical account by Morgan (1919) describes the presence of a limestone belt beginning a few kilometres southeast of the mountain sacred to Ngāi Tūhoe, Maungapohatu, and running for kilometres ('many miles') to the northwest. Morgan (1919) considered the Maungapohatu limestone deposit to be part of the Tertiary (Cretaceous) rocks found in the Tairāwhiti (Gisborne) and Hawke's Bay regions, and thus might extend in the northeast direction through the Huiarau and Kahikatea Ranges to the southern part of Raukūmara Range. Limestone deposits can contain naturally occurring phosphates (Thomas and Crutchfield 1974). These sedimentary deposits also include mudstones ('siltstones') that might also be a natural occurring source of phosphates (Porder and Ramachandran 2012). Pleistocene sandstone and siltstones are found south of Awakeri to Tāneatua and west of Ruatoki, as well as south of Ōhope as far as Waimana and east as far as the Raukokore River south of Cape Runaway. Past and present erosion in these regions due to the steep topography results in generally shallow soils over angular, shattered greywacke (Rijkse and Guinto 2010). It is possible that sedimentary facies (limestone, mudstone) in the predominately greywacke basement rock are contributing to elevated DRP in this part of the BOP. Alternatively, deposits of ejected and wind-blown tephra (e.g., pumice, ash) from volcanic eruptions might also be natural sources of P in the eastern BOP (Figure 2). Such geological features could help to explain why some of the predominantly forested catchments in the eastern BOP demonstrate elevated levels of DRP. For instance, the Ōtara River has a 5-year median DRP concentration of ≈0.03 mg/L (Figure 7) despite having a catchment predominately covered in native forest (86%) over a basement of greywacke as the dominant rock type.

Table 3. Mean and median concentrations (± 95% confidence intervals) of dissolved reactive phosphorus (mg/L) in Bay of Plenty streams and rivers. Sites have been grouped into categories based on catchment geology described in the River Environment Classification (REC) database. Land cover indicates broad categories assigned in the REC database. Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz), REC classifications for catchment geology from the MfE data portal (www.data.mfe.govt.nz).

Group	Land cover	Sites	Mean	Lower CI	Upper CI	Median	Lower CI	Upper CI
Volcanic	Native veg.	9	0.014	0.006	0.021	0.012	0.000	0.021
	Exotic forest	7	0.038	0.015	0.062	0.025	-0.018	0.033
	Pasture	27	0.026	0.013	0.039	0.014	0.006	0.018
	Urban	0						
Hard sedimentary	Native veg.	7	0.018	0.012	0.025	0.018	0.012	0.024
	Exotic forest	0						
	Pasture	0						
	Urban	0						
Alluvium	Native veg.	0						
	Exotic forest	0						
	Pasture	1	0.009			0.009		
	Urban	0						

However, although naturally sourced DRP is abundant in BOP streams and rivers, there also remain anthropogenic sources that require management. One challenge moving forward is knowing how much of the DRP load is naturally derived as opposed to sourced from human activities, and what might be useful approaches to help set more appropriate instream concentrations and exceedance criteria. Based the work by the BOPRC thus far, setting tighter nutrient limits on streams flowing into sensitive receiving environments (i.e., lakes and estuaries) should be prioritised over streams flowing directly into the ocean. The logic behind prioritising the downstream effects of DRP is because most streams in the BOP region are unlikely to display adverse responses of high nutrient concentrations (Snelder et al. 2019, Kilroy et al. 2020).

Streams in the BOP that flow from catchments dominated by volcanic geology (e.g., ignimbrite) and receiving geothermal inputs have an abundance of DRP which is naturally occurring and thus unlikely to respond to management interventions, except where receiving environments are also considered. The threat of elevated DRP concentration causing excessive periphyton growth is likely diminished by the underlying substrate types and propensity for "scouring" by frequent high-flow events ("freshs") in most areas of the BOP (Dare 2019). This observation is supported by the results of Kilroy et al. (2020) who showed a weak influence of total nitrogen on periphyton communities with a more pronounced effect of flood frequency, substrate size and temperature. DRP had no influence on observed periphyton biomass: indeed, its inclusion in the statistical models actually decreased the goodness of fit (Kilroy et al. 2020). It should be noted that that the streams used in the analyses by Kilroy et al. (2020) were selected to be as stable as possible, meaning that many of the streams in the central part of the BOP region not sampled were all SB, where the chance of periphyton development was even less. Dare (2019) estimated that overall, 20 of the 45 NERMN sites (44%) are unlikely to support blooms of periphyton due to the highly mobile substrate found at the sites. This assumption should be further validated to better understand if streams characterised by beds comprising of highly mobile fine sediment and pumice potentially mitigate the threat of periphyton exceeding trigger values.

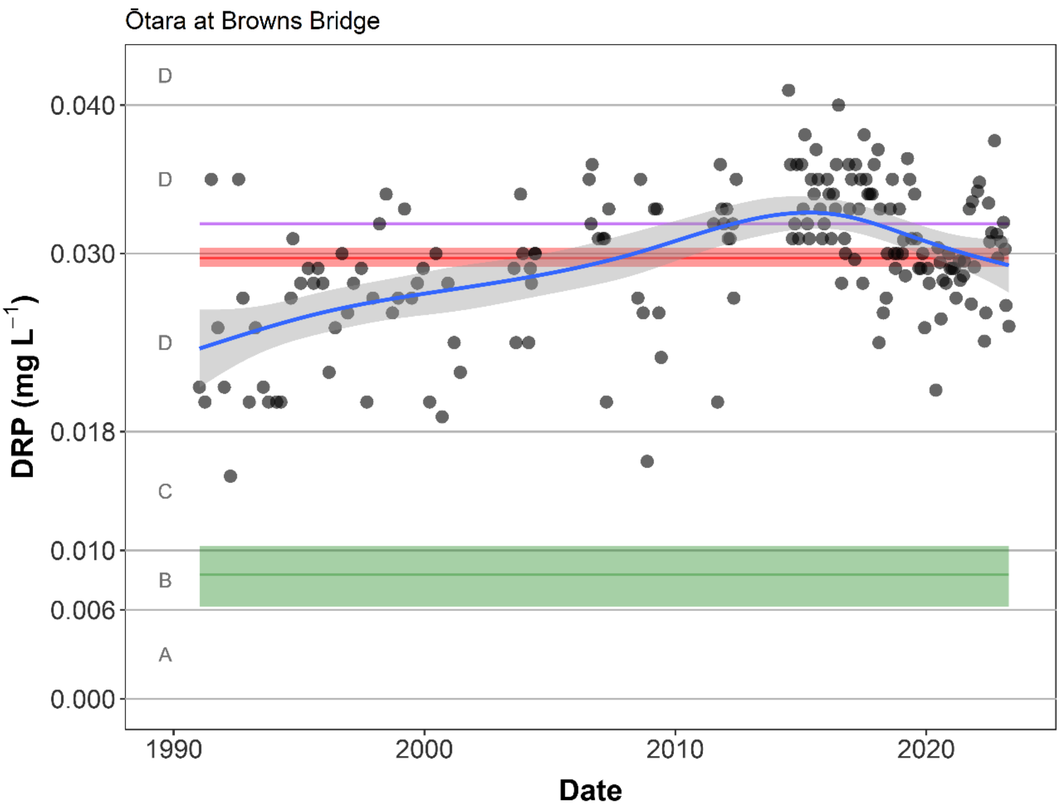

Figure 7. The Ōtara River in the eastern Bay of Plenty has shown persistently high concentrations of dissolved reactive phosphorus (mg/L) for the past 30 years, despite having a catchment dominated (86%) by native forest land cover. Plotted data fall within the 95% confidence interval to exclude outlier values (*n*=2). Attribute bands (A-D) under the NPS-FM are indicated. The blue line indicates a Generalised Additive Model (GAM) regression fit with the 95% confidence interval (CI) shaded in grey. The GAM fitted with the form *y ~ s(x, bs = "cs")* was highly significant (*p*<0.001) and explained 18.7% of the deviance. Trends in time may be due to increased erosion associated with Cyclone Bola (1988), DRP regeneration lags, and subsequent revegetation (Table 1). The median value (≈0.03 mg/L) for the last five years of monitoring in the Ōtara River is shown by the red line (red shading indicates the 95% CI). The green line indicates the median concentration of DRP found in North Island streams and rivers categorised as having catchments dominated by native vegetation (green shading indicates the 95% CI). This median value sits below the observed 80th percentile trigger value (0.010, 0.005-0.016 95% CI) estimated by McDowell et al. (2013) for the REC climate by topography by geology grouping (Cold-Extremely Wet-Lowland-Hard Sedimentary) that the Ōtara River occupies. The purple line indicates a potential trigger value scaled from the McDowell et al. (2013) estimate. This value is included as a proof of concept and is not intended for managing DRP in the Ōtara River. For more information on trigger values, see section 'Potential numeric attribute scores for DRP' (pg.20) and Appendix D. Ōtara River DRP data sourced from the BOPRC Environmental Data Portal (www.envdata.boprc.govt.nz/Data).

Clause 3.13(2) of the NPS-FM requires that where there are nutrient-sensitive downstream receiving environments, the instream concentrations and exceedance criteria, or the instream loads, for nitrogen and phosphorus for the upstream contributing water bodies must be set to achieve the environmental outcomes sought for the downstream environments. Thus, active management may be required in exceptional circumstances, such the aluminium dosing of the Utuhina and Puarenga Streams that discharge into Lake Rotorua. This management intervention helps to reduce DRP in each stream to concentrations below the limit of detection (McIntosh 2012). Generally speaking, active management is not a preferred option, however. This is because of potential ecological side-effects, along with concerns about how feasible, economic, or sustainable such interventions are over the long term. These realities help justify why exemptions due to naturally occurring processes are required for some streams and rivers in the BOP region.

Potential numeric attribute scores for DRP

A trigger value is broadly defined as a concentration of a contaminant that, if exceeded, alerts water managers to a potential negative change in ecosystem health and thus “triggers” a management response. As a precursor to the NPS-FM, the Australian and New Zealand Environment and Conservation Council (ANZECC) water quality guidelines were used extensively throughout New Zealand in managing water quality and potential effects of discharges. These guidelines used trigger values for a wide range of contaminants including DRP, which was set at 0.009 mg/l for lowland rivers, and 0.006 mg/l for upland rivers based on National Rivers Water Quality Network (NRWQN) datasets (Davies-Colley 2000). Attempts were made to identify baseline or pseudo-baseline (minimally or lightly impacted) sites to establish such triggers values, but the underlying analyses did not consider the yet to be developed River Environment Classification (REC) classes (other than excluding rivers with alpine influences). More recently McDowell et al. (2013) derived trigger values using statistical models to estimate reference conditions and critical values for appropriate classes defined by the second (climate and topography) and third (climate and topography and geology) levels of the hierarchical River Environment Classification (REC) developed by Snelder and Biggs (2002). This approach used appropriate reference information and REC categories to define trigger values for DRP using the 80th percentile of observed concentrations, given DRP is deemed to be harmful at high values (McDowell et al. 2013).

Developing regional trigger values (i.e., a numeric value for DRP that indicates an adverse change in attribute status) could be based on the methods described by McDowell et al. (2013). We have scaled the 80th percentile values from McDowell et al. (2013) to the median values for DRP concentrations in the BOP over the past five years to provide a ‘proof of concept’ for region-specific trigger values (Appendix D). It would be difficult to replicate this approach without the nationally derived data because BOPRC do not have sufficient reference sites in their sampling network. Nonetheless, the trigger-value approach provides options for the council when deciding how to deal with NOP *and* human activities that jointly contribute to elevated DRP concentrations in BOP streams and receiving environments.

Applying for an exemption to the NPS-FM for DRP makes sense in the presence of NOP (Table 1). However, the council should also consider an ongoing evidence-based approach using region-specific triggers values like those shown in Appendix D. Validating ecological harm caused when such potential trigger values are exceeded is contingent on appropriate indicators. Suitable bioindicators would demonstrate adverse ecosystem responses to excessive P from anthropogenic sources (discharges from wastewater treatment plants, effluent ponds, etc.). Attributes linked to DRP concentrations under clause 3.13(1) of the NPS-FM such as periphyton have shown little to no statistical responses to nutrients in the BOP region (Kilroy et al. 2020). This result does not mean that adverse effects of anthropogenic P are irrelevant to streams in the BOP region. The attributes prescribed under the NPS-FM are not exhaustive (e.g., functional indicators for decomposition could be considered), multiple-stressor interactions could cause unwanted 'ecological' surprises (Matthaei et al. 2010, Jackson et al. 2016), and sensitive downstream receiving environments are particularly at risk from eutrophication.

One argument against developing regional trigger values based on the approach used by McDowell et al. (2013) is that the REC classes used are inappropriate for the BOP region. McDowell et al. (2013) derived trigger values for the second and third levels of the hierarchical REC database, and it seems likely that the third level (Geology) is inadequate for the Bay of Plenty. The REC describes a range of natural factors influencing water quality (e.g., climate, source of flow and geology). It is widely used for understanding water quality patterns in New Zealand (Larned et al. 2005) but is not without limitations (Snelder et al. 2004). One of the major limitations of the current REC is the accuracy of the geology layer. The deficiencies in the REC geology layer explain why BOPRC have developed their own biophysical classification based on newer and more accurate geological maps. There may be room for further improvement in describing and understanding the geology of the eastern BOP. Other information sources, such as the "soft-volcanic" substrate/geology category according to the FWENZ spatial model used by Dare (2019), groundwater residence times and contributions to surface waters (Morgenstern et al. 2015), and mapping of geothermal inputs may prove more useful for accurately describing local reach-scale conditions.

Conclusions

Naturally occurring processes (NOP) clearly impact dissolved reactive phosphorus (DRP) concentrations in many streams and rivers of the Bay of Plenty (BOP) region. We have identified these NOP (e.g., geology and geothermal activity) that influence concentrations of DRP in these streams and rivers (Table 1). The region has unique geological features that leads to naturally occurring high concentrations of DRP in streams and rivers. Rock weathering in volcanic geologies and subsequent enrichment of groundwater and other receiving environments, along with geothermal inputs and putative sedimentary sources of P likely explain instances where streams and rivers in the BOP region achieve "D" attribute status for DRP, especially in catchments with extensive native forest cover (i.e., relatively low anthropogenic influence). Alongside these NOPs, we have indicated the likely increase in attribute state (i.e., to the B-D bands) depending on the mechanisms driving elevated DRP in streams and rivers (Table 1). These insights are important for BOPRC when considering appropriate target attribute states (TAS) and policies relating to improving DRP in the region. Alternative attribute state tables for DRP in the BOP could be based upon the trigger values provided in Appendix D, although more consideration is required for this approach. In some instances, the best available knowledge is still insufficient to fully understand the putative natural sources of P indicated in Table 1 (i.e., in the eastern BOP).

Lake Dissolved Oxygen (Bottom and Mid-hypolimnetic)

Context

All aquatic animals need dissolved oxygen to breathe. Dissolved oxygen is the amount of oxygen that is present in water. Lakes predominantly receive oxygen from the atmosphere and aquatic plants, including phytoplankton. A reduction in dissolved oxygen concentration is generally attributed to eutrophication: the process of a shift in the trophic state in aquatic ecosystems from oligotrophic to more eutrophic conditions. These trophic state changes are characterised by nutrient concentrations (i.e., nitrogen and phosphorus) and primary production (algal or macrophyte biomass).

The breakdown of organic matter (e.g., phytoplankton biomass settling through the water column) consumes oxygen in the water column and at the sediment-water interface. Increased oxygen demand in the bottom waters can lead to reduced habitat availability and quality for biota, including fish, mussels, and kōura, resulting in changes in food web dynamics (Pingram et al. 2021). In extreme cases, oxygen-devoid water can lead to mass fish deaths. During periods of anoxia, a series of biogeochemical processes lead to the release of phosphorus and nitrogen from the bottom sediment into the overlying water column.

Dissolved oxygen (DO) attribute in the NPS-FM

Two attributes related to dissolved oxygen concentrations are defined in the NPS-FM: lake-bottom dissolved oxygen (NPS-FM Table 18) and mid hypolimnetic dissolved oxygen (NPS-FM Table 19). These tables provide a description for the impacts of dissolved oxygen changes on lake ecological communities and biogeochemical functions in the hypolimnion and bottom waters. While such an approach is important for setting policy objectives and implementing measures for resource use and management within environmental limits (Johnson 2013), the implication that changes to lake ecological communities and biogeochemical processes will occur and respond predictably at exact thresholds for these attributes, set nationally, is debatable and neglects that other, usually un-monitored factors can result in adverse effects on ecological communities. This is furthermore complicated by the important role that other internal nutrient loading mechanisms (not related to oxygen mediated nutrient releases) have in many lakes, in a complex interplay between physical, chemical, and biological interactions in lake ecosystems.

Dissolved oxygen is typically measured directly in the lake. BOPRC carries out monthly profiling of dissolved oxygen in 11 Rotorua Te Arawa lakes, and high frequency autonomous monitoring stations located in seven lakes. Scholes (2021) reviewed the adequacy of the current monitoring scheme to report against the dissolved oxygen attributes in the NPS-FM and noted that existing monitoring does not meet the lake bottom monitoring requirements. Furthermore, the monthly monitoring frequency in polymictic lakes may misrepresent the actual attribute state based on comparisons between the high frequency monitoring data with the monthly data.

Hypolimnetic oxygen depletion in stratified water bodies can be assessed using indicators such as volumetric hypolimnetic depletion rate (VHOD) and areal hypolimnetic oxygen deficit (AHOD). These indicators offer estimates of the rate at which dissolved oxygen is depleted in the lower layer of the water body. However, they do not capture the full extent of anoxic conditions in terms of both spatial and temporal variations within lakes. Therefore, it is crucial to measure and report dissolved oxygen concentrations as well, as they directly impact other dynamics related to the health of the ecosystem.

Current status of dissolved oxygen in the Rotorua Te Arawa lakes

Scholes (2021) calculated the minimum lake dissolved oxygen reading for the year 2019/20 and found that six out of 11 lakes did not meet the national bottom line for this attribute, despite detailed commentary on the adequacy of the monitoring programme. Most of the lakes appear to conform to the generalisable relationship between eutrophication and bottom water dissolved oxygen; that is, the more eutrophic a lake is the more likely bottom water anoxia will occur (Müller et al. 2012). This relationship is applicable to monomictic and polymictic lakes, although detecting anoxia in polymictic lakes using monthly profile data can be challenging. In this study, mid-hypolimnion dissolved oxygen concentrations calculated from monthly profile data for eight monomictic lakes show that seven lakes did not meet the national bottom line for this attribute. Only values for Lake Tarawera were just above the national bottom line at 4.17 g/m^3. There is a strong correlation between minimum lake dissolved oxygen concentration attribute and the mid-hypolimnion dissolved oxygen concentration attribute for monomictic lakes (R^2=0.85, p<0.05) suggesting that both attributes are influenced by similar processes and can be discussed in the same context of NOP.

Mechanisms of the deoxygenation of the hypolimnion in lakes

There are basically three factors that contribute to the deoxygenation of the hypolimnetic waters of lakes. One is the biochemical oxygen demand of the planktonic algae produced in the epilimnion of lakes (autochthonous organic material) and organic matter transported from the catchment to the lake (allochthonous organic material), which sink into the hypolimnion. Another is the oxygen demand of lake bottom sediments. The third is certain chemicals, such as ammonia and biodegradable organics, developed within the lake or derived from the watershed. The latter is not further discussed in this report as our literature review has revealed a paucity of systematic research on this topic. There is considerable debate in the literature surrounding the relative significance of these factors and the influence of other factors on the deoxygenation of hypolimnia, which are discussed below for the Rotorua Te Arawa lakes.

Importance of allochthonous vs autochthonous sources of organic material

Organic carbon (in dissolved and particulate form) from catchments contributes to lakes by several pathways. The two primary catchment sources are vegetation, via leaching of litter and live vegetation, and soils, via microbial metabolism, root exudation, and leaching and erosion of soil organic matter.

Historically, the sources of organic carbon in the catchments of the Rotorua Te Arawa lakes would have been derived from tall indigenous forests that have covered the Rotorua Lakes Ecological District (McEwen 1987). Tawa was the most common forest tree, with various podocarps (predominantly rimu) and northern rātā scattered throughout. Some variants of this forest type would have occurred on land below 300 m a.s.l. in the northeast where kohekohe, pūriri and nīkau were also present. Organic carbon sources around Lake Rotorua would have been derived from wetlands that occurred extensively around the lake. Typical wetland plant species included kahikatea, harakeke, raupō, swamp coprosma, and sedges. Specialised vegetation types included shrublands that were associated with geothermal areas, lakeshores, and areas of recent volcanic activity.

The above background is important for understanding the sources and fate of organic carbon to the lakes in the context of NOP, as substantial remnants of mainly logged podocarp-hardwood forests occur mainly on higher ground (predominantly rimu/tawa, kāmahi, pukatea, mangeao). Only small areas of hard beech in gorges on the edge of the Mamaku plateau remain along with remnant groves of kahikatea; inland pōhutukawa and some other coastal species round shores of lakes. Specialised vegetation close to sites of geothermal activity remain intact in some places, characterised by prostrate kānuka and frost tender ferns with tropical affinities. The 1886 eruption of Mt. Tarawera that resulted in much early successional vegetation, and bare scoria fields of the mountain summit (Timmins 1983).

The implications of the above for assessing NOP for the deoxygenation of the hypolimnion in lakes is implicit in that the lake catchment areas currently covered in indigenous vegetation only partially reflects the organic carbon source of what would have occurred under natural conditions. Furthermore, the modification of the landscape for sheep and cattle farming, dairying, horticulture, and exotic forestry has very likely altered the nature and mixture of organic carbon sources to the lakes.

McBride et al. (2020) estimated the proportion of current natural and anthropogenic land use in the catchments of the lakes and the associated loads of nitrogen and phosphorus for each catchment. The proportion of natural land use in the catchments range from 9.2% (Lake Ōkaro) to 83.4 % (Lake Ōkataina) with an average across all catchments of 49.1% and a standard deviation of 23.4% (Table 4). Unfortunately, organic carbon sources from the catchments to the lakes have not been quantified. Effects of mixed land use on carbon transport varies substantially between catchments, as delivery rates can change due to different agricultural and forest management practices (e.g., Moore 1989). This author measured dissolved organic carbon (DOC) exports form eight Maimai catchments (North Westland) and found that 8-10 years after clear-cutting, concentrations, and export of DOC from the clear-cut catchments are 1.2-2.4 times higher than in two undisturbed native forest catchments. This difference was especially pronounced in catchments where the slash had not been burnt. Similarly, Bright & Mager (2016) found that particulate organic carbon (POC) export was higher in a forested catchment which was undergoing clearance compared to a catchment covered in tussock and woody scrub in the Glendhu Experimental Catchments.

While the organic carbon export from non-natural land use might be higher compared to natural land use, the fate of organic matter with regards to the reactivity and composition (and thus its effect of oxygen demand) can be contrariwise. For example, Young and Huryn (1999) studied the effects of land use on metabolism and carbon transport in five streams in the Taieri River catchment in the southeast of the South Island. They found that despite relatively low amounts of material transported to the stream with a native forest catchment, respiration (oxygen consumption) was higher compared to other streams in forested and grassland catchments, which showed higher transported organic matter. The findings above suggest that the reactivity and mineralisation rates of organic matter derived from native vegetation could lead to a disproportional effect on oxygen consumption in the receiving water bodies compared to organic matter derived from non-native land use. However, we have no evidence for corroborating this assertion for the Rotorua Te Arawa lakes and it is noted that the extreme complexity of natural organic matter and the diversity of its interactions with the environment make it difficult to identify the principal drivers of organic matter degradation as our understanding our of how changes in organic matter quality translate into changes in decomposition rates (and therefore oxygen consumption rates) is incomplete.

Research from mostly northern hemisphere temperate zone lakes suggest that the organic matter mineralisation in lakes from autochthonous dominates over allochthonous organic carbon pools. For example, autochthony was the dominant source of organic carbon respiration in the hypolimnia of six northern-temperate Wisconsin lakes (Delany et al. 2023). Organic matter mineralisation from autochthonous organic carbon pools had higher turnover rates than allochthonous organic carbon pools in the oligotrophic Rappbode Reservoir in the Harz Mountains, Germany (Dordoni et al. 2022). However, autochthonous organic matter was found in lower concentrations than the more recalcitrant allochthonous material in 39 lakes of the northern highland region of Wisconsin and

Michigan. (Wilkinson et al. 2013). In a New Zealand study, Saeed (2022) characterised the evolution of dissolved organic matter (DOM) in the water column of Lake Ngāpouri, located in the Waikite Valley. This author argued that the observed humic-like allochthonous carbon inputs associated with high-intensity summer rainfall displayed increasing photochemical oxidation rates because terrestrially derived DOM is known to absorb significant amounts of solar radiation. Thus, temporal variability of the relative importance of mineralisation of autochthonous vs allochthonous organic material may contribute to this contrasting finding along with complex chemistry of metal oxides in this lake that can have a strong impact on mineralization of organic matter (Saeed 2022).

Sediment oxygen demand

The oxygen demand in lake bottom sediments is primarily driven by the organic matter content in the sediments, which can contribute substantially to the deoxygenation of the hypolimnetic waters. Because the production of organic matter either in the watershed or in the water body itself is the ultimate route for organic carbon production and deposition to the bottom sediment it is important to understand the likely sources of sediment organic matter.

Previous sediment carbon analysis for the Rotorua Te Arawa lakes showed that water column chlorophyll *a* concentration has a significant positive relationship with areal storage of carbon in the sediment for the period 1886 to 2006 (Santoso et al. 2017). The deposition rates of carbon ranged from 0.31 to 2.82 mol C/m^2/year, with eutrophic lakes having higher deposition fluxes than the oligotrophic lakes. These results indicates that phytoplankton productivity (related to anthropogenic eutrophication) is the dominant process contributing to carbon deposition in the sediment of the Rotorua Te Arawa lakes. This is further supported by a strong relationship (R=0.81, p<0.01) found between sediment carbon content in the lakes and percent catchment area in agriculture (Bruesewitz et al. 2011), and a study by Trolle et al. (2008) showing that the carbon to nitrogen ratio (C/N: 4.8 – 10.1) of the surficial sediment organic matter in these lakes is predominantly sourced from an autochthonous base (C/N<10; Meyers, 1994).

Lake morphology

Lake morphology can substantially influence the processing of organic matter in the hypolimnion of lakes. For example, maximum depth is generally well correlated to lake volume, which is important in oligotrophic lakes, because deeper, more voluminous monomictic lakes hold a larger reserve of oxygen and can better mitigate its loss in the hypolimnion before the autumnal mixing event replenishes oxygen to the entire water column. Likewise, in eutrophic lakes, the extended residence time of organic particles in deeper hypolimnia allows for more thorough oxidation, as they have longer exposure to the oxygen available. In shallower lakes, organic particles settle quickly and become part of anoxic sediments, where they serve as a substrate for microbial activity that primarily relies on less efficient anoxic respiration.

The thickness of the hypolimnia (which can change during a single stratification period) determines the time available that sedimenting organic particles spend to fully oxidised (consume more oxygen). In deep hypolimnia compared to shallow hypolimnia of equal volume, sedimenting organic particles have more opportunity to oxidise because of longer contact with oxygenated hypolimnetic water during sedimentation. In shallow hypolimnia, the particles become more quickly part of the mostly anoxic sediments where they serve as a substrate for microbes. Because anoxic respiration is less efficient, it ultimately consumes less oxygen than if these particles had spent more time in a deeper, better oxygenated water column.

The relative area of a lake under the top of the metalimnion relates to oxygen depletion by providing a measure of the areal extent of the lake sediment area in contact with water below the epilimnion. In lakes with large shallow areas where much of the lake sediment is in contact with epilimnetic waters (small area under metalimnion), oxygen loss from decomposition of sediment organic matter may be largely replenished from surface mixing in the epilimnion. In contrast, if this shallow area of the lake is proportionally small (large area under metalimnion), the larger isolated portion of the lake will undergo oxygen consumption without epilimnetic replenishment.

Groundwater inputs into the Rotorua Te Arawa lakes can be substantial (McBride 2020; Wilson 2022). Depending on the relative contribution and the location of groundwater inputs in the overall water balance, groundwater could affect lake bottom water dissolved oxygen dynamics. Our literature review has revealed no targeted study (neither on the Rotorua Te Arawa lakes or lakes elsewhere) that would allow an in-depth discussion of groundwater effects on hypolimnion oxygen dynamics. Using a heuristic argument, it can be reasoned that, for example, groundwater inputs into Lakes Tarawera and Rotomāhana could affect hypolimnion oxygen dynamics by some degree. This argument is based on the water balance for these lakes calculated by Wilson (2022) who used best estimates of data and flows to calculate the percentage contributions of average annual flows to and from these lakes, including groundwater. These estimates suggests that groundwater inputs are roughly equivalent to 11% and 17% of the hypolimnetic water volume in Lakes Tarawera and Rotomāhana, respectively, assuming a stratification period of eight months in these lakes. The actual effect on hypolimnetic on lake bottom water dissolved oxygen dynamics are highly uncertain and will depend on the location of entry and temporal variation of groundwater inputs and the oxygen concentrations within the groundwater.

Elevated temperatures can stimulate microbial activity, resulting in increased rates of organic matter decomposition and increasing oxygen demand. This effect of temperature maybe an important consideration for the Rotorua Te Arawa lakes, of which several have geothermal heat inputs (e.g., McBride et al. 2020). However, our calculations of volumetric average temperature of the hypolimnia in the lakes suggest that at this scale (whole hypolimnion), elevated temperature effects on microbial activity may be negligible. Localised effects, however, where geothermal activity occurs cannot be discounted.

Quantifying drivers of hypolimnetic oxygen demand in the Rotorua Te Arawa lakes

One approach that has been found useful in identifying those factors which have a major impact on hypolimnetic oxygen depletion for a given waterbody was developed in the US OECD Eutrophication Study (Rast and Lee 1978). These studies produced empirical regressions between planktonic algal biomass in epilimnetic waters and the rate of oxygen depletion in hypolimnetic waters. Other researchers have continued to develop empirical relationships to describe the hypolimnetic oxygen depletion rate as a function of epilimnetic chlorophyll concentration. The models that were developed describe the monthly, volumetric, hypolimnetic oxygen depletion rate as various functions of average, annual chlorophyll concentration modified by various combinations of morphological characteristics of the waterbody. Similar patterns were found in four South Island New Zealand lakes by Schallenberg and Burns (1999) who demonstrated the dependence of areal hypolimnetic oxygen depletion rates on epilimnetic chlorophyll *a* concentration.

Charlton (1980) developed a simple regression model between hypolimnion oxygen demand (expressed as AHOD) and lake productivity, hypolimnion volume, and temperature (Table 4). The models in that study were tested on a comparably small number of North American Great Lakes but resulted in high correlation between observed and calculated hypolimnetic oxygen demand. To illustrate the effects of productivity, hypolimnion depth and temperature on expected minimum hypolimnetic oxygen concentration in the Rotorua Te Arawa lakes, equation 11 in Charlton (1980) is illustrated in Figure 8. By solving the equation for various combinations of chlorophyll *a* concentrations and metalimnion thickness, an impression of the magnitude of response can be gained. The figure shows the minimum concentration of oxygen in the hypolimnion of a lake calculated after 100 days of stratification at different values for metalimnion thickness and temperature, and chlorophyll *a* levels, with values comparable to those observed in the Rotorua Te Arawa Lakes (Table 5). The optimisation of the equation parameters for the Rotorua Te Arawa Lakes results in a moderately good fit of calculated AHOD values (*R*=0.58, *RMSE*=0.97), which must mean that the assumptions about the importance of productivity, hypolimnion depth and temperature apply for the Rotorua Te Arawa Lakes. However, a large amount of variation in AHOD values remain unexplained using this approach, which suggests that other unquantified drivers of oxygen consumption play a role in the lakes, including sediment oxygen demand and mineralisation of allochthonous organic matter (as discussed above).

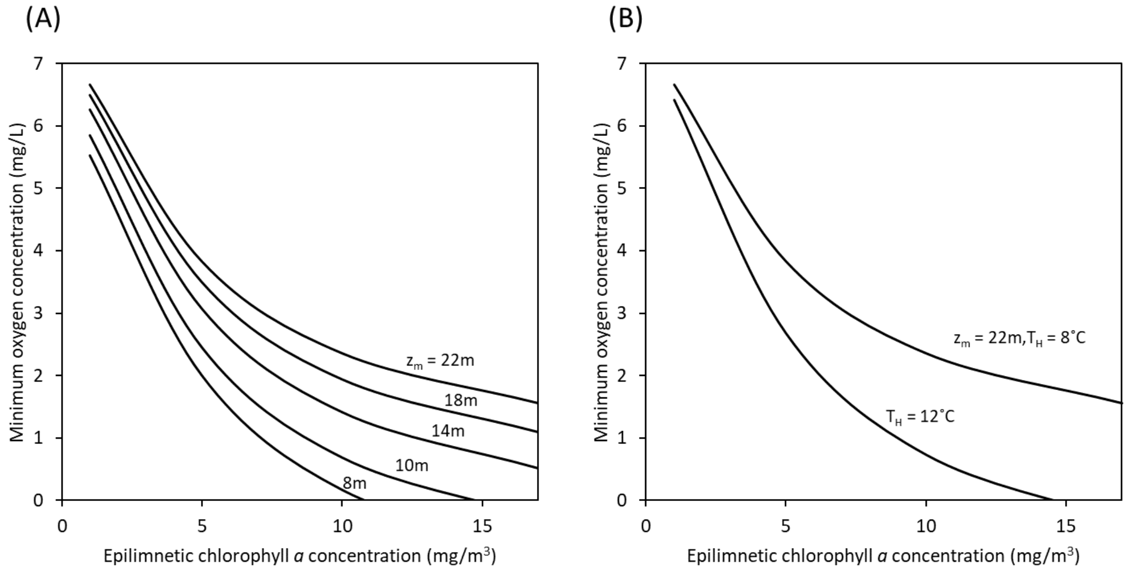

Figure 8. Effects of (A) hypolimnion thickness and (B) hypolimnion temperature, and different epilimnetic chlorophyll *a* concentrations on minimum hypolimnion oxygen concentrations using equation 11 from Charlton (1980). The value range for hypolimnion thickness and chlorophyll *a* concentration is based on ranges comparable to the Rotorua Te Arawa lakes. The assumed duration of stratification was 100 days with an initial oxygen concentration of 11 mg/L.

Quantitative interpretation of Figure 8 and the calculated AHOD values must be done with caution as the model parameters have not been completely validated for this extrapolation. A logical conclusion following from the above premise is that oxygen depletion rates correspond with the degree of eutrophication and by implication can be translated into catchment nutrient loads if load - response relationships are known. However, variations (seasonal and inter-annual) in lake and hypolimnetic depth, temperature, and internal processes complicate the direct assessment of nutrient load reductions on bottom water oxygen dynamics.

Nevertheless, in the absence of any other measurements of organic carbon sources contributing to the oxygen demand in the lakes, we have calculated the ratio of the estimated reference AHOD and the estimated current AHOD to provide an indicative proportion of naturally occurring oxygen demand based on epilimnetic productivity (i.e., allogenous to autochthonous organic matter). We used estimated reference concentrations of chlorophyll *a* for the Rotorua Te Arawa lakes derived by Abell et al. (2020) and applied these in the Charlton (1980) equation, while keeping all other variables constant. The ratio of the reference AHOD and current AHOD should therefore reflect the potential naturally occurring oxygen demand as a function of increased eutrophication in the lakes (i.e., the manageable component of oxygen demand through nutrient load reductions), while acknowledging that organic matter derived from allochthonous sources are reflective of current land use. The mean ratios ranged from 0.34 (i.e., 34% for Lake Rotoiti) to 1.21 (i.e., 121% for Lake Rotomā; Table 5). The average ratio was 0.81 with a standard deviation of 0.33. Low values of this ratio indicate that a large proportion of the current oxygen demand in a given lake may be due to anthropogenic eutrophication, while large values indicate that current oxygen demand may be largely due to naturally occurring processes, since a key driver of oxygen demand (i.e., epilimnetic productivity) was set to its reference condition for a specific lake. Ratio values > 1 are most likely indicative of the inherent uncertainty with this approach (see Table 4 for estimated uncertainty), albeit the current chlorophyll *a* concentration in the oligotrophic Lakes Ōkataina (mean ratio of 0.98), Rotomā (1.21), Tikitapu (0.99) and Tarawera (1.12) are near reference condition. There was an intuitively strong correlation between the Trophic Level Index and the AHOD ratio (R=0.85, p<0.01), which indicated that the more eutrophic a lake is, the higher the proportion of anthropogenic contributions to AHOD. Likewise, there was a strong relationship between the proportion of catchment in natural condition and the AHOD ratio (R=0.48, p<0.05), highlighting that lakes with nearly pristine catchments exhibit AHOD values close to their reference condition.

Illustrating the case of oligotrophic lakes using the example of Lake Ōkataina

Few studies have examined the link between natural causes and deep-water oxygen depletion in oligotrophic lakes. However, lakes free from extensive catchment development can exhibit seasonal anoxia because of morphometry and acidic dissolved organic carbon exported from catchment vegetation and wetland areas (Nürnberg 2004). Lake basin morphometry may be a primary driver of both dissolved oxygen depletion rates and the areal extent of anoxia in some low-nutrient lakes. For example, Deeds et al. (2021) analysed the occurrence of natural vs. anthropogenically influenced anoxia in 414 low-nutrient lakes (total phosphorus <15µg/L) in Maine, United States of America. By including 36 morphological variables in their analysis, they found that the areal extent of the lake sediment area in contact with water below the epilimnion was amongst the strongest predictors for anoxic events in these lakes. They argued that in lakes with large shallow areas where much of the lake sediment is in contact with epilimnetic waters, oxygen loss from decomposition of sediment organic matter may be largely replenished from surface mixing in the epilimnion (Table 4). In contrast, if this shallow area of the lake is proportionally small, i.e., a large proportion of the lake surface area below the epilimnion, the larger isolated portion of the lake will undergo oxygen consumption without epilimnetic replenishment. The value range of areal extent of the lake sediment area in contact with eplimnetic water in Deeds et al. (2021) was 49.29 – 79.43%. By comparison, the proportion of the area under the top of the metalimnion in the Rotorua Te Arawa Lakes is 70.50 – 99.02% (Table 5) and suggests that these lakes are susceptible to anoxic conditions in the hypolimnion based on their morphological characteristics.

Table 4. Naturally occurring processes (NOP) that can cause low lake bottom dissolved oxygen concentrations in the Rotorua Te Arawa lakes. Arrows indicate direction of change from attribute A status (i.e., going up to B-D status).

Naturally Occurring Process		Mechanism	Key references	Likely occurrence/ importance in the BOP	Likely effect on bottom and mid-hypolimnion DO attributes band (relative to A status)
Trophic state	Epilimnetic chlorophyll *a* concentration	Dependence of areal hypolimnetic oxygen depletion rates on epilimnetic chlorophyll *a* concentration (i.e., phytoplankton biomass). Some lakes have chlorophyll *a* concentration close to reference condition.	Schallenberg and Burns (1999), Burns et al. (1999).	High	↑ C-D
Lake morphology	Lake depth	In oligotrophic lakes: deeper, more voluminous monomictic lakes hold a larger reserve of oxygen and can better mitigate its loss in the hypolimnion before the autumnal mixing event replenishes oxygen to the entire water column. In eutrophic lakes: the extended residence time of organic particles in deeper hypolimnia allows for more thorough oxidation, as they have longer exposure to the oxygen available.	Deeds et al. (2021)	Medium-High	↑ B-C
	Thickness of hypolimnion	Sedimenting organic particles spend more time and are more fully oxidised (consume more oxygen) in deep than in shallow hypolimnia of equal volume because of longer contact with oxygenated hypolimnetic water during sedimentation.	Cornett and Rigler (1987)	Medium-High	↑ B-C
	Relative area of lake under top of metalimnion	The relative area of lake under top of metalimnion relates to oxygen depletion by providing a measure of the areal extent of the lake sediment area in contact with water below the epilimnion.	Deeds et al. (2021)	High	↑ B-D
Thermal conditions	Volumetrically weighted average hypolimnetic temperature	Elevated temperatures can stimulate microbial activity, resulting in increased rates of organic matter decomposition and increasing oxygen demand, particularly for lakes with geothermal heat sources.	Charlton (1980)	Medium	↑ B

An example of anoxia in a low-nutrient might be Lake Ōkataina: a deep oligotrophic lake where a large proportion of the hypolimnion becomes anoxic during the stratified period (Kpodonu 2016). The lake is predominantly surrounded by indigenous vegetation (81% of the catchment), with the remainder comprising exotic pine plantations (8%) and dry stock farming (c. 11%). The Trophic Level Index value for the lake was 2.6 in 2019/20 (Scholes 2021) and can be considered near pristine based on the estimated mean reference TLI for volcanic lakes of 2.8 (Abell et al. 2020). Scholes (2021) argued that the near pristine TLI in Lake Ōkataina could render the dissolved oxygen attribute unhelpful for target setting, since the management of nutrient inputs into the lake might not result in an improved trophic state. Scholes (2021) asked: "How are water quality practitioners meant to interpret the minimum dissolved oxygen in the context of achieving environmental outcomes, particularly if an improved state might still mean the lake remains under the national bottom line?".

During stratification, the average thermocline depth in Lake Ōkataina is 18.2m (Table 5), which would be equivalent of approximately 83% of the lake surface area below the epilimnion on average. It is therefore reasonable to conclude that the lake morphology of Lake Ōkataina (and other lakes based on values presented in Table 5) renders the lake susceptible to naturally occurring anoxic events. Interestingly, prior to 1980, the hypolimnion of the lake had been reported to be completely oxygenated even during thermally stratified periods (Jolly 1968, McColl 1972). Kpodonu (2016) used a range of paleolimnological methods and time series analysis to determine long-term trends of water quality in Lake Ōkataina. He found that recent impacts of invasive mammals and climate warming tended to act synergistically to increase catchment loads of sediment and nutrients to the lake, effectively creating a shifting reference condition for a lake that would otherwise be considered relatively stable. Anthropogenically derived nutrient and sediment transport from the catchment into the lake cannot be discounted, and excessive erosion caused by feral ungulate and macropod species (Kpodonu 2016) might explain the lack of obvious links between land use and lake water quality in Lake Ōkataina.

Conclusions

To ensure accurate attribute states for dissolved oxygen in the Rotorua Te Arawa lakes, it is crucial to have appropriate monitoring data as described in Scholes (2021) as a starting point. While monthly measurements of vertical oxygen profiles can be used to determine dissolved oxygen concentration attribute states in lakes that undergo seasonal mixing (monomictic lakes), this sampling frequency is insufficient for lakes that experience frequent mixing (polymictic lakes). In the case of polymictic lakes, the monitoring buoys currently operated by BOPRC offer enough data to capture the dynamics of dissolved oxygen. In this context, it is argued that the assessment of dissolved oxygen attributes based solely on concentration has limited value in enhancing our understanding of the processes that contribute to the occurrence of anoxic bottom water conditions. To gain insights into the factors affecting oxygen dynamics in the bottom water, it is more meaningful to calculate hypolimnetic oxygen depletion rates as complementary attributes to the NPS-FM attributes. The specific method for calculating these rates, AHOD or VHOD, depends on the intended purpose. These rates can be correlated with the amount of biodegradable organic material present in the lake, while also considering other factors that influence bottom water oxygen dynamics. In this regard, designing a study to better understand the sources and fate of different organic matter compounds for the Rotorua Te Arawa lakes and their catchments would shed light on how lake ecological processes interact with external forcing that led to the development of anoxia.

The approach taken in this report shows that hypolimnetic oxygen dynamics in the Rotorua Te Arawa lakes are partially driven by primary production in the surface waters, and by association trophic state driven by land use activities (a manageable component). This relationship is well established for these lakes (Scholes and Hamill 2016). Natural causes of anoxic conditions cannot be discounted in all the lakes based on the exploration of general relationships between hypolimnion oxygen demand, lake productivity, hypolimnion volume, and temperature.

Several NOP impact lake bottom and mid-hypolimnetic DO concentrations in the Rotorua Te Arawa. The likely effects of NOP on dissolved oxygen bands are outlined in Table 4. All of the lakes have morphological characteristics that renders them susceptible to low concentrations of lake bottom and mid-hypolimnetic DO. This susceptibility is further highlighted by our calculations of reference oxygen demand (using AHOD), with oligotrophic lakes (in catchments with extensive native forest cover) showing similar reference AHOD values to current AHOD values. This important finding highlights the need to consider NOP when considering appropriate TAS and policies relating to improving lake dissolved oxygen in the region. Based on our findings, we have highlighted very indicatively potential TAS for lake bottom and mid-hypolimnetic DO attributes for all lakes in Table 5.

It is important to note that hypolimnetic oxygen depletion and anoxia in productive lakes can be mitigated by reducing autochthonous production of organic carbon, which is mainly driven by anthropogenic land use and nutrient loads. This argument is true for all Rotorua Te Arawa lakes (even for oligotrophic lakes) because all lakes have an appreciable amount of anthropogenically derived nutrient input (McBride et al. 2020). If nutrients are managed sustainably, epilimnetic productivity (a key driver for oxygen consumption in bottom waters) should decrease.

Table 5. Summary of Rotorua Te Arawa Lakes characteristics discussed in this document. A more detailed description of variables can be found in the main body of the text.

Lake name	Mixing pattern	Trophic state	Minimum bottom water oxygen concentration (mg/L)[1]	Trophic Level Index[2]	Chlorophyll *a* (µg/L)[2]	% lake area under top of metalimnion	Current AHOD ($g/m^2/day$)[3]	Reference AHOD ($g/m^2/day$)[4]	Potential TAS for bottom and mid-hypolimnion DO attributes
Ōkāreka	Monomictic	Mesotrophic	0.13	3.1	3.1	79.08%	0.49	0.36 (0.27-0.48)	B-C
Ōkaro	Monomictic	Eutrophic	0	4.6	11.9	83.07%	0.57	0.28 (0.22-0.36)	C
Ōkataina	Monomictic	Oligotrophic	0.15	2.6	1.9	83.00%	0.40	0.39 (0.28-0.55)	C[5]
Rerewhakaaitu	Polymictic	Mesotrophic	0.1	3.7	4.9	-	-	-	B-C[6]
Rotoehu	Polymictic	Eutrophic	0.6	4.7	17.1	-	-	-	B-C[6]
Rotoiti	Monomictic	Eutrophic	0.52	4.1	8.4	77.18%	0.99	0.33 (0.24-0.52)	C
Rotokākahi	Monomictic	Mesotrophic	-	3.5	3.8	-	-	-	-
Rotomā	Monomictic	Oligotrophic	3.21	2.4	1.2	70.50%	0.30	0.37 (0.26-0.54)	B-C
Rotomāhana	Monomictic	Mesotrophic	5.95	3.6	3.4	99.02%	0.60	0.37 (0.27-0.53)	B-C
Rotorua	Polymictic	Eutrophic	0	4.2	10.9	-	-	-	B-C[6]
Tarawera	Monomictic	Oligotrophic	4.3	2.7	1.5	91.31%	0.36	0.41 (0.28-0.59)	B-C
Tikitapu	Monomictic	Oligotrophic	0.19	3	2.5	80.91%	0.40	0.39 (0.28-0.53)	C-D

[1]Scholes (2021); Minimum dissolved oxygen readings combined for discrete vertical profiles (for monomictic lakes) and lake buoy sensors (for polymictic lakes). Lake Rotokākahi values were omitted due to lack of data.
[2]Hamill (2022); Trophic Level Index values and chlorophyll *a* concentrations represent the average for the years 2019 – 2021.
[3]Values were calculated as VHOD x average metalimnion depth based on unpublished VHOD values from BOPRC. Lake Rotokākahi values were omitted due to lack of data.
[4]Values were calculated using reference chlorophyll *a* concentrations derived by Abell et al. (2020). Lower and upper bounds in parenthesis indicate 95% confidence intervals based on confidence intervals for reference chlorophyll *a* concentrations
[5]Lake Ōkataina has a history of well oxygenated bottom waters (see main text).
[6]Potential TAS for bottom and mid-hypolimnion DO attributes for polymictic lakes will require further consideration as the majority of NOP discussed in this document may only be applicable to monomictic lakes.

Deposited Fine Sediment

Context

Deposited fine sediment (DFS) is mud, silt or sand on the bed of the stream or river. It exists naturally in streams, rivers and other waterbodies as part of the hydrofluvial system. However, rapid land clearance following European colonisation and ongoing impacts of intensive human land uses in New Zealand have greatly increased fine sediment inputs to streams, rivers and other receiving water bodies. Excessive DFS can have wide-ranging impacts on stream ecosystems by removing benthic habitat necessary for a range of aquatic life and functions, including the macroinvertebrates used for State of the Environment (SOE) biomonitoring (Burdon et al. 2013).

Deposited Fine Sediment (DFS) attribute in the NPS-FM

One of the 12 NOF attributes in the NPS-FM requiring an action plan when values are below bottom lines is DFS (Table 16 in Appendix 2B of NPS-FM but provided below in Figure 9). In Figure 9 the indicator score is the percentage of cover of the streambed in a run habitat, determined by an assessment of fine sediment deposition using the SAM2 methodology of Clapcott et al. (2011). Here fine sediment is taken to be inorganic particles that are less than 2 mm in grain size, thus incorporating fine sand, silt, and mud. Bands for these attributes are based on a median of 80 visual assessments taken over five years of monthly monitoring (or longer where flow conditions only prevent seasonal monitoring). Four different river classes have also been recognised (see Appendix A in this document) in an attempt to recognise the inherent differences between rivers of different climate, source of flow, and geology based on the River Environment Classification (REC) database.

Current status of DFS in the BOP

Suren (2020) recognises the pervasive impact of fine sediment deposition on stream ecosystems. This view reflects the extensive work done in different parts of New Zealand highlighting the deleterious effects of DFS on biodiversity and ecosystem functioning in streams and rivers (e.g., Matthaei et al. 2010, Wagenhoff et al. 2012, Burdon et al. 2013). More recently, Snelder et al. (2019) highlighted the influence of DFS on key macroinvertebrate metrics in the Bay of Plenty. However, Suren (2020) takes issue with the DFS attribute in NPS-FM (Figure 9). There are five main concerns outlined by Suren (2020) with the implementation of the DFS attribute under the NPS-FM. These concerns are summarised below as questions:

1. Is the approach to manage DFS appropriate for the Bay of Plenty?
2. Does monthly sampling provide more information than annual sampling?
3. Are small differences in attribute bands ecologically meaningful?
4. Do DFS attributes selected for the NOF have an influence on biotic metrics?
5. Will dynamic hydrogeomorphic processes associated with restoration affect the TAS?

We have appraised these five concerns with reference to the NPS-FM and other literature whilst considering the broader issues of DFS before concluding with a final recommendation.

Table 16 – Deposited fine sediment

Value (and component)	Ecosystem health (Physical habitat)			
Freshwater body type	Wadeable rivers			
Attribute unit	% fine sediment cover			
Attribute band and description	**Numeric attribute state by deposited sediment class**			
	Median			
	1	2	3	4
A Minimal impact of deposited fine sediment on instream biota. Ecological communities are similar to those observed in natural reference conditions.	≤7	≤10	≤9	≤13
B Low to moderate impact of deposited fine sediment on instream biota. Abundance of sensitive macroinvertebrate species may be reduced.	>7 and ≤14	>10 and ≤19	>9 and ≤18	>13 and ≤19
C Moderate to high impact of deposited fine sediment on instream biota. Sensitive macroinvertebrate species may be lost.	>14 and <21	>19 and <29	>18 and <27	>19 and <27
National bottom line	**21**	**29**	**27**	**27**
D High impact of deposited fine sediment on instream biota. Ecological communities are significantly altered and sensitive fish and macroinvertebrate species are lost or at high risk of being lost.	>21	>29	>27	>27
The indicator score is percentage cover of the streambed in a run habitat determined by the instream visual method, SAM2 as defined in p. 17-20 of Clapcott JE, Young RG, Harding JS., Matthaei CD, Quinn JM. and Death RG. 2011. *Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values*. Cawthron Institute: Nelson, New Zealand. (*see* clause 1.8) Based on a monthly monitoring regime where sites are visited on a regular basis regardless of weather and flow conditions. Record length for grading a site based on 5 years. *See* Tables 24 and 26 in Appendix 2C for deposited sediment classes and their composition. This attribute does not apply in river environment classes shown in Table 25 in Appendix 2C, or where clause 3.25 requires freshwater habitat monitoring.				

Figure 9. Table 16 from Appendix 2B of the National Policy Statement for Freshwater Management (NPS-FM) 2020 outlining the attribute bands for deposited sediment.

Question 1: Is the approach to manage deposited sediment appropriate for the Bay of Plenty?

One of the key concerns raised by Suren (2020) is the meaning of a degraded state with regards to the attributes and respective bands described in the NPS-FM (Figure 9). Here *degraded* relates to a Freshwater Management Unit (FMU) or part of an FMU, so that because of something other than a NOP the attribute (e.g., DFS) is below an NBL, or is not achieving or is unlikely to achieve a TAS. From Table 16 (Figure 9): the DFS "attribute does not apply in river environment classes shown in Table 25, Appendix 2C of the NPS-FM (Appendix A in this document), or alternatively where clause 3.25 requires freshwater habitat monitoring". The REC groups in Table 25 (Appendix A) are freshwater habitats where DFS is deemed to be naturally occurring. These are limited in their applicability, and as such we have developed a broader view of naturally occurring processes (NOP) leading to elevated DFS in streams and rivers with special regard to the Bay of Plenty (Table 6).

Table 6. Naturally occurring processes (NOP) that can cause elevated levels of deposited fine sediment (DFS) in stream and rivers. The areas that are likely affected by NOP in the Bay of Plenty (BOP) region are indicated, including River Environment Classification (REC) groups based on Climate, Source of Flow, and Geology. Feral mammals in the BOP include wild populations of Dama wallaby (*Macropus eugenii*), along with red deer (*Cervus elaphus*), fallow deer (*Dama dama*), pigs (*Sus scrofa*) and goats (*Capra hircus*). Arrows indicate direction of change from attribute A status (i.e., going up to B-D status).

Naturally Occurring Processes		**Mechanism**	**Key references**	**Where in the BOP**	**Likely occurrence/ importance in the BOP**	**Likely effect on DFS attribute band (relative to A status)**
Climate	Drier conditions	Reduced vegetation cover can lead to increased erosion, and decreased flows reduces assimilative capacity of stream to process sediment	Baldan et al. (2021)	High rainfall in the region means this mechanism is unlikely to play a major role in DFS, although climate variability and irrigation/municipal demands should be considered (e.g., Katikati)	Low	↑ B-C
	Wetter conditions	Extreme rainfall may lead to landslides and bank erosion, increasing sediment supply and exceeding assimilative capacity of stream	Neverman et al. (2023)	Increasingly extreme weather involving high rainfall in the region mean this mechanism could have acute impacts on areas of the BOP with steep topography (e.g., Kaimai and Raukūmara Ranges)	Medium	↑ B-D
Source of Flow		Low gradients and stable flows mean fine sediment can naturally accumulate with low assimilative capacity through reduced transport	Clapcott et al. (2011), Burdon et al. (2013), Konrad and Gellis (2018), Dingle et al. (2020)	Lowland streams, Lake outlets (e.g., Ōhau Channel, Tarawera River), and streams with spring sources (e.g., Hamurana Springs)	Medium-High	↑ B-D
Geology		Eroding layers of fine geological material can increase sediment supply and lead to naturally high levels of DFS	Wallace and Benke (1984), Collier and Halliday (2000), Collier and Bowman (2003), Collier and Smith (2003)	Alluvial deposits in lowland areas, areas where volcanic geologies dominant, particularly areas with deep layers of volcanic ash and pumice (e.g., Rangitaiki River catchment including the Kaingaroa Plateau, Mamaku Plateau, etc.; see Figure 2)	High	↑ B-D
Biological	Riparian shading	Riparian trees on banks formerly under pasture can lead to bank readjustment and sediment erosion	Davies-Colley (1997), Quinn et al. (1997)	Pastoral sites undergoing riparian restoration or conversion to forest (Exotic, Native)	Medium	↑ B-C
	Feral Mammals	Reduced vegetation cover and greater erosion leading to increased sediment supply	Scanes et al. (2021)	Forest (Exotic, Native) with volcanic and/or sedimentary soils, lower order headwater streams	Low	↑ B

A key feature of the BOP region that leads to NOP affecting DFS (Table 6) is the extensive distribution of volcanically derived, pumice-dominated soils (Figure 2). This tephra layer includes fine pumice, often many metres deep, broadly deposited over volcanic and non-volcanic basement rocks in the region. This material is a legacy of the extensive volcanism that shaped and continues to influence the geology and hydrogeomorphology of the region (Lowe 1990). The historic eruptions deposited fine pumice and ash over much of the region, which have slowly been cut into by streams and rivers as they work their way through this easily erodible material. This NOP helps explain why so many hard-bottomed HB streams (according to Table 24 in the NPS-FM) may actually be soft-bottomed (SB).

This problem is relevant when applying the NPS-FM. The situation envisaged in clause 3.25(2) of the NPS-FM is where a site previously characterised as soft-bottomed is determined to be naturally hard-bottomed, thus requiring a specific set of management actions. Suren (2020) observes that clause 3.25(1) states: "If a site to which a target attribute state for DFS applies (Figure 9) is soft-bottomed, the *regional council must determine whether the site is naturally soft-bottomed or is naturally hard-bottomed*." The same table also states that "this attribute does not apply in River Environment Classification (REC) classes presented in Table 25", which lists the REC groups that are naturally soft-bottomed (SB). Suren (2020) interprets this to mean that all other REC classes (presented in Table 24 of the NPS-FM but see Appendix A) are in fact hard-bottomed (HB).

The main concern Suren (2020) thus poses is if the REC classes outlined in Tables 24 and 25 are appropriate to determine if a site is naturally soft-bottomed (SB) or is naturally hard-bottomed (HB). The REC classes used in the NPS-FM are likely mischaracterising SB streams as HB in the BOP region. Using a different methodology to describe DFS than that mandated by the NPS-FM 2020 (i.e., SAM2), Suren (2020) presents evidence which suggests a strong discrepancy between field conditions in the Bay of Plenty and the classifications according to the REC classes as outlined in Tables 24 and 25. According to the NPS-FM, Suren (2020) reports all but one of the 149 sites currently monitored by the BOPRC as part of the SOE programme should be classified as HB. This is in stark contrast to the descriptions using the SAM3 method, which indicates that 69 sites are field defined as HB and 80 are SB (> 50% of the streambed classified as sand or finer). These observations are evidence that some sites in the Bay of Plenty may have naturally occurring processes meaning current and baseline states are below NBLs and thus TAS will need to be below national bottom lines. However, to establish this with certainty we first need to investigate the potential differences between methods used to describe DFS and why the REC classes might be inadequate for accurately characterising the stream types in the Bay of Plenty, before suggesting a path forward.

The method described by Suren (2020) appears to be the SAM3 protocol in Clapcott et al. (2011). This protocol is also known as the Wolman pebble count and is a semi-quantitative assessment of the particle size distribution including DFS on the streambed. At least 100 particle measurements are made within a single site. The method relies on the "Wolman walk" method of randomly selecting substrate particles on the streambed (Wolman 1954) and recording the b-axis length (perpendicular to the longest axis) of each particle using a modified Wentworth scale (Wentworth 1922). This method contrasts with the SAM2 protocol which relies on visual assessments of DFS (<2 mm) on the streambed. The SAM2 protocol is also semi-quantitative and relies on 20 points within a site each assessed at four grids in the underwater viewer (e.g., a bathyscope). The two methods (SAM2 and SAM3) for describing DFS are highly correlated. Clapcott et al. (2011) reports correlations of r=0.677 (p<0.01, n=82) and r=0.68 (p<0.01, n=167) between the two protocols. Although the SAM3 protocol provides important information about benthic substrates other than DFS, there are indications that it underestimates % sediment cover. In Clapcott et al. (2011) the SAM3 values typically fall below the 1:1 with the SAM2 estimates of sediment cover. In our experience, it can be harder to perceive the fine sediment deposited between larger benthic particles using the SAM3 method, which might help explain why it appears to underestimate sediment cover. Another estimation error might come from differences in interpreting what constitutes DFS: if there is a thin layer of fine sediment (e.g., silt) on top of larger substrates (e.g., cobbles), observers should record this as "silt", but consistently applying this approach may be difficult in practice. Results from a survey of 25 streams in the wider Te Awanui / Tauranga Harbour catchment confirm these observations. There was a strong correlation between the two protocols for estimating %DFS, but with a systematic bias in the SAM3 protocol to underestimating % sediment cover (Figure 10).

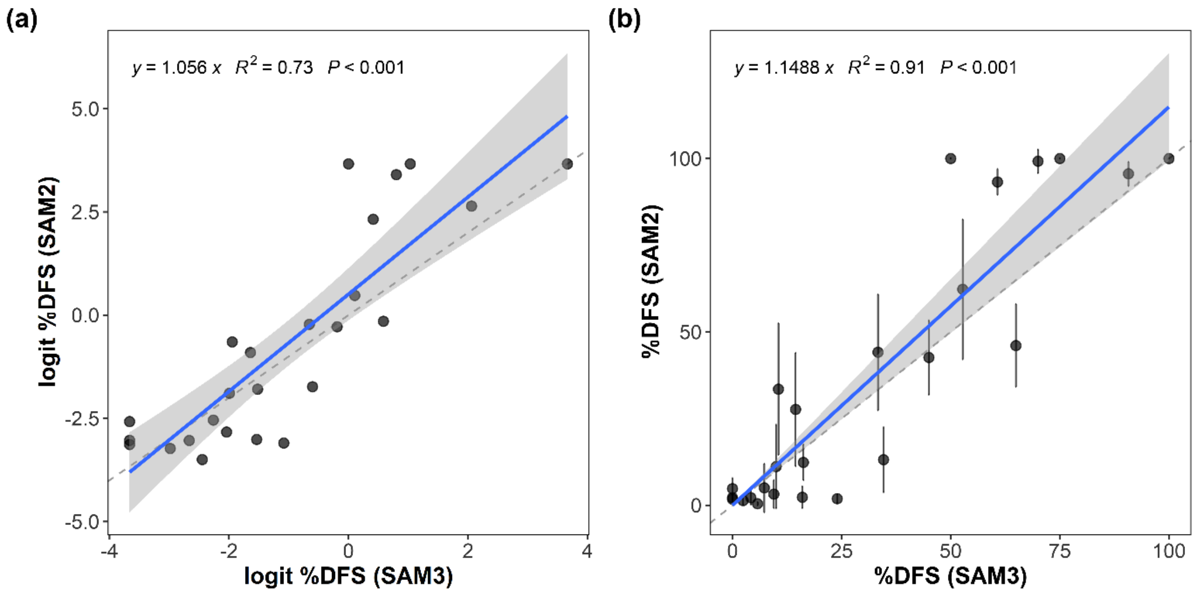

Figure 10. Estimates of deposited fine sediment (DFS) are highly correlated using the SAM2 (In-stream visual assessment of sediment) and SAM3 (Wolman pebble count) protocols. Data are from 25 streams sites in the wider Te Awanui / Tauranga Harbour catchment recorded by University of Waikato MSc student Brooklyn Lea (Unpublished data). (a) The logit transformation corrects for the beta distribution of the proportion DFS data to reveal a highly correlated relationship between the two protocols. (b) The untransformed values indicate that %DFS using the SAM2 protocol (as prescribed in the NPS-FM) can be predicted from %DFS using the SAM3 protocol (*x*-intercept = 0). Point error bars indicate the 95% confidence interval for %DFS estimated using the SAM2 protocol, grey shading indicates the 95% confidence interval for the linear regressions. The 1:1 line is shown by the dashed line.

However, although the methods may alter the accuracy of DFS cover estimates, this does not detract from the main argument of Suren (2020) that the REC classes that identify HB streams based on their Climate/Source of Flow/Geology classification appear to inadequate for accurately characterising stream types in the Bay of Plenty. Whether a stream is naturally dominated by fine sediment is dependent on a number of factors including stream size, catchment vegetation, rainfall, slope, and geology. Although the NPS-FM recognises that some lowland rivers with gentle slopes and weak stream power (i.e., lack the ability to mobilise sediment) are naturally SB (see Table 25 but Appendix A here), other stream types may be SB as the result of naturally occurring processes (Table 6). For example, Clapcott et al. (2011) describes that very small streams with gentle slopes and low rainfall on sandy soils can be naturally dominated by fine sediment substrates. SB streams may currently account for approximately 20% of the length of rivers in New Zealand, according to Freshwater Ecosystems of New Zealand (FENZ) classification (Leathwick et al. 2011). In contrast to the FENZ classifications, predictions from models using River Environment Classification (REC) data suggest less than 2% of all NZ streams would have greater than 50% fine sediment cover in the absence of human land-use activities (Clapcott et al. 2011). The discrepancy between these two estimates suggests that there are potentially major challenges in using modelled outputs to determine SB and HB status in streams as presented in Tables 24 and 25 of the NPS-FM (Appendix A here). Thus, solely relying on determinations of stream type derived from REC data and subsequently Table 25 in the NPS-FM is likely to be misleading, particularly in the BOP region. This is because it is predicated on the information available at the time of determination, and at any rate, even updated databases may remain insufficient to deal with the complexity of the natural environment (e.g., volcanic geologies) in the Bay of Plenty.

The REC database contains catchment spatial attributes, summarised for every segment in New Zealand's network of rivers. Individual river sections are mapped according to physical factors such as climate, source of flow for the river water, topography, and geology, and catchment land cover (e.g., forest, pasture or urban). This information is mapped for New Zealand's entire river network (425,000 kilometres of river) and enables sites to be categorised based on their ecological similarities. However, Snelder et al. (2004) reported that the classification strength of the REC was low. The low classification strength of the first four levels of the REC (Climate, Source of Flow, Geology and Landcover), which describe variation in watershed characteristics, was due to the importance of local-scale factors such as substrate and hydraulic habitat in explaining spatial variation in invertebrate communities. The results of Snelder et al. (2004) suggest that although the REC provides a general representation of ecological patterns, it is unable to reliably predict characteristics for a given site because it cannot account for local habitat or flow heterogeneity.

Thus, using the REC to characterise stream types could be prone to error, and may help explain why Suren (2020) observes such large discrepancies in realized vs predicted conditions in the BOPRC NERMN monitoring sites. The other factor to consider is the nature of the streams themselves. Streams in the Central North Island region which extends north into the Bay of Plenty (e.g., the Rangitaiki River catchment) are typically spring-fed with benthic substrate dominated by mobile beds of pumice sand and gravels (Collier and Halliday 2000). Geologies of the region typically include a mantle of tephra including volcanic ashes and pumice gravels underlaid by welded igneous rocks (Collier and Bowman 2003, Collier and Smith 2003). The soils include weakly structured yellow-brown pumice which is prone to severe sheet erosion. Such unique features suggest that the REC approach may incorrectly characterise the streams as hard-bottomed, and that naturally occurring processes contribute to the relatively high levels of DFS in the BOP region. For instance, 108 of the 128 (84%) streams used by the BOPRC for macroinvertebrate biomonitoring are described as Volcanic. The streams with beds dominated by tephra ash and other fine inorganic substrates seemingly possess geologies that would be better described in the Soft Sedimentary REC Geology group, yet this category seems to be seldomly assigned in the georeferenced REC database. A 'Plutonic Volcanic' sub-group exists within the Soft Sedimentary category, yet this classification is inappropriate to describe the tephra layers that are a feature of the central North Island. This oversight of the REC database is a strong argument for the BOPRC to seek exemptions under the NPS-FM for the DFS attribute as they develop their own more robust classification systems for the Geology group in the REC database.

However, care will be needed when assessing if catchment land-use practices are appropriately managed in such SB streams since they may be predisposed to additional sedimentation issues (e.g., highly erodible banks with sediment inputs from livestock intrusion and forestry activities). Correctly determining these streams as naturally SB does not exclude best management practices, and land-use impacts on other attributes (e.g., suspended sediment) may mean that limits on resource use and/or action plans should be enacted in accordance with the NPS-FM. Sediment runoff from surrounding catchments can have adverse effects on sensitive downstream receiving environments. In the case of Tauranga Harbour, upstream erosion can have direct and indirect effects on the estuary including sedimentation, loss of seagrass, and spread of mangroves (Hume et al. 2010). Moreover, climate change models predict rainfall intensity will increase in most parts of the BOP region, which will further amplify sediment runoff (Parshotam et al. 2008). This modelling work predicts that catchment sediment runoff to Tauranga Harbour will increase by 40% by 2051, with effects greatest in pastoral agriculture. Similar predictions have been obtained from catchment modelling for the entire BOP region, with the increase in total erosion by mid-century assuming median climate change projections with contemporary landcover ranging from 24% to 83% (Vale et al. 2021). Best practice methods may arrest these changes by 6.6% (Vale et al. 2021), although additional restoration (e.g., riparian afforestation) may have disproportionate benefits to sensitive downstream receiving environments by addressing erosion and providing greater resilience to climate change (Witing et al. 2022).

Suren (2020) recommends that DFS monitoring be restricted to sites that have been classified by field surveys as HB using the SAM3 protocol. Whilst I can see the logic behind this recommendation, it is inherently problematic since it relies on the *a priori* assumption that excessive sediment is naturally sourced. This approach may be useful to address current and future sediment issues in HB sites but could also ignore present and persistent sediment problems in sites incorrectly designated as SB streams. Validating the assignment of SB or HB status should start with the best available knowledge using the most up-to-date and accurate geological maps for the region developed by the BORPC. However, the opportunity remains to build superior environment classification models in the future that incorporate other information about environmental conditions at these sites such as the degree of hydrogeomorphic degradation, sediment depth and loads, stream gradients, and the past and present intensity of land use in the upstream catchment. Such an approach would provide the optimal understanding of past, present and future conditions. Sediment depth could be particularly useful and was trialled as a potential metric to assess naturally SB streams in national protocol developments (Clapcott et al. 2011). It was not developed further because at the national level it was poorly related to invertebrate biotic metrics and was the only metric not correlated with other measures of DFS. Sediment depth data was not as readily available as other protocol data (Clapcott et al. 2011). Suren (2020) also recommends that the classifications given in Tables 24 and 25 (based on REC) to determine if a site is HB are ignored. We can see value in BOPRC developing their own classification based on the best available information and thus support this recommendation by Suren (2020).

Question two: Does monthly sampling provide more information than annual sampling?

Suren (2020) questions the value of monthly sampling to describe DFS. Whilst onerous there is evidence that historic datasets based on sediment protocols including in-stream visual methods (e.g., SAM2) can be used to detect significant trends in sediment over time (Clapcott et al. 2011). Such temporal variation in sediment may be caused by seasonal flow influences, or sediment pulses as a result of land use or natural disturbance events. DFS data from 11 years of monitoring of streams on the Whatawhata Research Station farm in the Waikato Region showed that fine sediment was nearly 30% higher in March than in September (Quinn et al. 2009). In another study highlighted by Clapcott et al. (2011), Quinn and Vickers (1992) reported significant temporal patterns along the hard-bottomed Tongariro River. Sediment varied markedly with season and tended to be higher in winter and lower in spring. This study indicates that DFS at a site might fluctuate greatly within a year, adding considerable support to regular and frequent monitoring of this attribute. Furthermore, since DFS monitoring under the NPS-FM relies on the median value from five years of data collection, collecting more, and not less site information will help determine the true extent of sedimentation at SOE sites. However, I can also imagine that at least some of the spring-fed streams present in the volcanic plateau areas of the Bay of Plenty likely exhibit less temporal dynamism, and if these streams are determined to be naturally soft-bottomed then monthly monitoring would no longer be required.

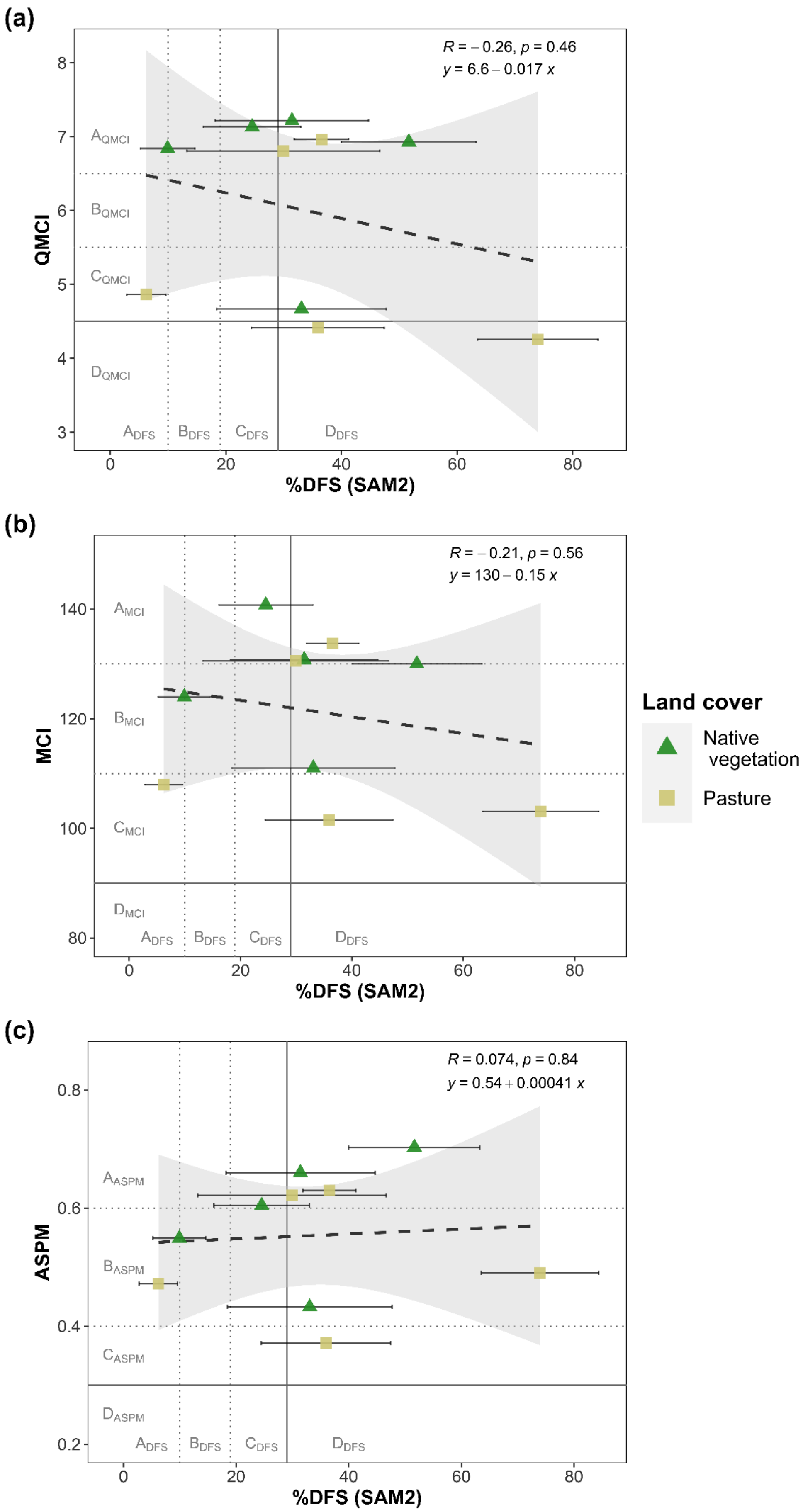
(a)
R = – 0.26, p = 0.46
y = 6.6 – 0.017 x
QMCI
A QMCI
B QMCI
C QMCI
D QMCI
A DFS
B DFS
C DFS
D DFS
%DFS (SAM2)
(b)
R = – 0.21, p = 0.56
y = 130 – 0.15 x
MCI
A MCI
B MCI
C MCI
D MCI
%DFS (SAM2)
(c)
R = 0.074, p = 0.84
y = 0.54 + 0.00041 x
ASPM
A ASPM
B ASPM
C ASPM
D ASPM
%DFS (SAM2)
Land cover
Native vegetation
Pasture

Figure 11. (*on previous page*) The absence of a statistically significant relationship between deposited fine sediment (DFS) and key macroinvertebrate indices: (a) QMCI; (b) MCI; and (c) ASPM indicates how naturally occurring processes can lead to finer particles dominating benthic substrate without a concomitant loss in stream ecosystem health. Ten sites were sampled by University of Waikato MSc student Shana Edgecombe in the Kuratau River catchment of the central North Island (Unpublished data). All sites are characterised by the River Environment Classification as CW-Hill-VA (thus Class 2 for DFS in Table 24 of the NPS-FM; see Appendix A). Point error bars show 95% confidence intervals for DFS. Vertical and horizontal lines indicate NPS-FM attribute bands. The dashed regression lines are all non-significant and the shaded 95% confidence intervals show the high degree of uncertainty in responses to DFS.

Question three: Are small differences in attribute bands between the sediment classes ecologically meaningful?

Suren (2020) challenges the ecological and statistical relevance of the relatively small differences between bands for the four streams classes in Table 16 (Figure 9). These classes are numbered 1-4 and represent different groups according to the REC (Table 24 in Appendix 2C of the NPS-FM, but see Appendix A here). A major concern is between class differences in attribute bands. For example, only a 6% difference in DFS estimates between the four classes exists for the A band, and only an 8% difference between the four classes in the D band. Furthermore, there is only a 2% difference in DFS in the D band between Classes 2, 3, and 4, and only a 2% difference in the C band between these three classes. The main argument by Suren (2020) is whether the small differences between the classes within each band are ecologically relevant, particularly if the classes themselves are flawed as identified earlier. These are valid concerns.

In defence of the NPS-FM, differences among classes should be taken in context of the median values from five years of data collection, where smaller differences between site types would be expected, albeit with less variation that one would see from a single year's sampling. This is based on the observation that as a sample size increases, the confidence in an estimate also increases, uncertainty decreases, and there is greater precision. Small differences may equate to large differences in ecological responses, such as the %EPT insects (Ephemeroptera, Plecoptera, Trichoptera) in the stream macroinvertebrate community. In relatively homogeneous gravel-bottomed streams of Canterbury, Burdon et al. (2013) found evidence for a threshold response to deposited sediment where the %EPT rapidly declined when deposited sediment exceeded 20% of the streambed over a 30-m reach. The response was scale-dependent: when considering the response at the patch scale (i.e., the area of the Surber sampler where macroinvertebrates were collected), the invertebrate response to DFS showed greater sensitivity with a threshold detected at approximately 13% (Burdon et al. 2013). However, fundamentally the same problem applies here – if the REC mischaracterises stream classes in the BOP region, then the small differences Suren (2020) highlights could be ecologically meaningful and lead to incorrect management actions.

Pumice-bed streams in the central North Island indicate that DFS can be much higher due to the naturally occurring processes (i.e., volcanic geology leading to the tephra layers of fine ash and pumice) without associated declines in macroinvertebrate community metrics (Figure 11). Ten sites were sampled for DFS and macroinvertebrates in the Kuratau River catchment of the central North Island. The Kuratau River is a tributary of Lake Taupō that flows through native forest and scrub, pine plantations and pastoral agriculture. Despite the low site replication, the results in Figure 11 are notable in the context of that study because at least four streams achieved "A" status on the macroinvertebrate attributes under the NPS-FM, but "D" status for DFS (>29% DFS). This suggests a disconnect between these attributes where naturally occurring fine particles dominate stream benthic substrate. All the sites in the Kuratau catchment are characterised by the REC database as *Cold-Wet* climate, *Hill* source of flow, and *Volcanic* geology, or CW-Hill-VA (thus Class 2 for DFS in Table 24 of the NPS-FM). The hill source of flow may ensure DFS is dynamic at the patch-scale due to steeper channel gradients and greater stream power. Dynamic flow conditions create a patch mosaic of different substrate patches (Burdon et al. 2013) which may help rheophilic EPT (Ephemeroptera, Plecoptera, Trichoptera) stream insects such as *Deleatidium spp.* mayflies to persist in streams of the Kuratau catchment. Woody riparian vegetation can also help to ensure that instream habitat heterogeneity remains high with submerged large wood providing a particularly important stable refuge (Collier and Halliday 2000) and geomorphic structuring agent (Quinn et al. 1997). Furthermore, low intensity land-use catchments like that of the Kuratau River may have relatively stable habitat dynamics at larger spatial scales, meaning they demonstrate weak or no relationships between invertebrate metrics and instream sediment levels (Davis et al. 2022). These factors help to explain the observed high scores for the MCI, its quantitative equivalent, and the Average Score Per Metric (APSM) indices in the four streams with high levels of DFS (Figure 11). The CW-Hill-VA group is well represented in the Bay of Plenty, with 21 (16%) of the SOE macroinvertebrate biomonitoring sites coming under this group.

We assessed the influence of four REC categories on macroinvertebrate communities over the past five years in the Bay of Plenty (BOP) using publicly available data (www.lawa.org.nz) and linear mixed-effects (LME) models. Three of the REC categories used also determine the four DFS classes specified in the NPS-FM. These REC categories are Climate, Source of Flow, and Geology. The fourth REC category included in our LME models was Land cover. We included a random effect for stream site nested in catchment to account for repeated measurements on the same sites and the nestedness of sites in the same river catchments. We first fitted a global model looking for additive effects of the four REC categories. This model ruled out climate as a having a significant influence on the MCI scores (Table B3). The strongest influence came from land cover in the final model (but see below regarding the interaction with source of flow). Native vegetation streams had significantly higher MCI scores than pasture or urban streams (both $p<0.05$). There was no significant difference between native vegetation and exotic forest streams, or exotic forest and pasture streams. Urban streams had significantly lower MCI scores than exotic forestry streams ($p<0.05$) but not pasture streams (Table B4). Regarding source of flow (Table B5), hill streams had significantly higher MCI scores than lowland ($p<0.01$) or lake-outlet streams ($p<0.05$). For the geology category, only the one 'alluvium' site differed from the other categories (Table B6), having considerably lower MCI scores compared to the volcanic, hard and soft sedimentary streams (all $p<0.01$) due to its highly degraded state on the coastal plains near Papamoa.

We also considered the interaction of land cover and source of flow (Figure 12) on MCI scores in the BOP region. This mixed model used the same random effect structure described above but indicated a considerable improvement in goodness of model fit when compared with the global model ($\Delta AIC = -13.9$). This model indicated that there was a significant interaction between land cover and source of flow (Table B7). The post-hoc tests indicated that for hill streams, there were no differences in MCI scores between land uses (Table B8). In contrast, lowland streams in forested areas (native and exotic) had significantly higher MCI scores than pastoral or urban streams (Figure 12). The MCI scores for lowland streams in native vegetation and exotic forest land covers did not differ. Lake-outlet streams with exotic forest cover had significantly lower MCI scores than the pastoral sites ($p<0.01$). The difference in MCI scores between lake-outlet streams in pasture and native vegetation was not statistically significant ($p=0.055$). MCI scores for lake outlet streams in native vegetation and exotic forest did not differ. Caution should be given when interpreting the results for the lake-outlet streams, since they represent single sites for forested land covers, and only two sites for pastoral land cover.

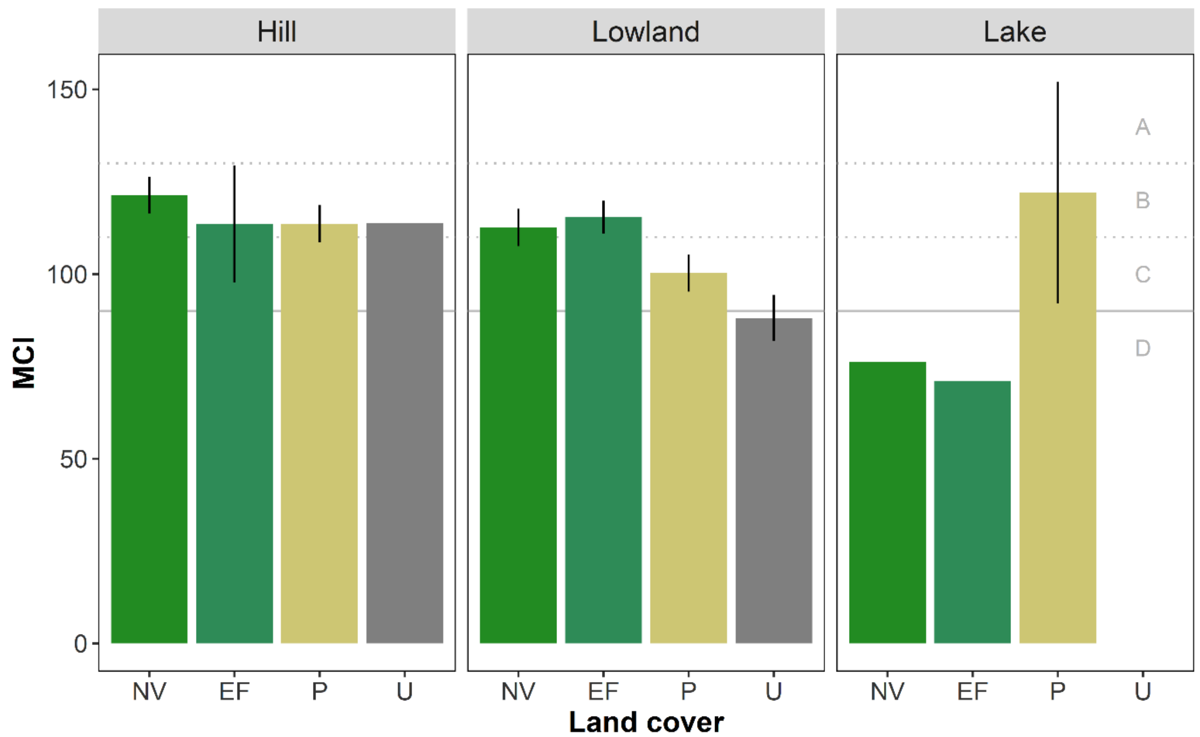

Figure 12. Mean (±95% confidence interval) Macroinvertebrate Community Index (MCI) scores in Bay of Plenty streams and rivers. Sites have been grouped into categories based on source of flow and land cover described in the River Environment Classification (REC) database. Land cover indicates broad REC categories (NV, Native vegetation; EF, Exotic forestry; P, Pasture; U, Urban). Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz), REC classifications for source of flow from the MfE data portal (www.data.mfe.govt.nz).

When compared to the rest of the North Island and the South Island, the MCI scores in the Bay of Plenty suggest that macroinvertebrates are not disproportionately impacted by the naturally high levels of fine sediment in the streams of this region (Table B2). The lack of differences indicates that other factors, such as the underlying intensity of land uses (Davis et al. 2022), presence of riparian vegetation, instream habitat heterogeneity, and source of flow might mitigate naturally elevated levels of DFS in the BOP region.

This is not to say that DFS is a minor management issue in the BOP; the results of Snelder et al. (2019) clearly highlight the importance of this attribute in explaining variation to macroinvertebrate community structure in the region. However, it is likely that the results of Snelder et al. (2019) also reflect wider issues relating to hydrogeomorphic and riparian degradation (Burdon et al. 2013), thus requiring further nuanced analyses to disentangle these co-occurring drivers. On this count, preliminary work is underway using data collected by the University of Waikato MSc student Brooklyn Lea. Our results here using publicly available MCI data do indicate that hill source of flow streams may be more resilient to naturally occurring DFS through increased assimilative capacity and greater habitat heterogeneity, provided the impacts of human land uses are not excessive. There is a need to better understand the temporal dynamics of DFS, since medium-intensity land-use catchments might exhibit greater variability in DFS at the reach scale, and thus cause greater impacts on invertebrate metrics since sediment-sensitive macroinvertebrates are able to persist in these catchments (Davis et al. 2022).

Suren (2020) also makes the comment that sites in native forest can demonstrate considerably large variability. Again, in context of the median values from five years of data collection for the NPS-FM, such fears may be unwarranted, provided the stream is not incorrectly characterised as SB. If natural processes account for elevated sediment levels and can be proven, then there should be a mechanism that allows for an exception to bypass the need to develop an action plan for what are effectively least-disturbed sites. The nature of pumice-bed streams in the Central North Island helps to underscore the need for this NOP report. Regarding concerns about sediment methods, Quinn et al. (2009) recorded elevated DFS in a forest stream using the SAM3 protocol after a tree fell into the reach and accumulated DFS upstream, demonstrating that sediment protocols can detect changes due to natural influences even at relatively small scales. However, this requires frequent monitoring to establish the impacts of natural processes. Furthermore, sediment inputs in native forest may not be directly related to human impacts, but *in theory* still require a management response. Feral mammals (e.g., goats, deer, pigs, horses) can have impacts on streams in nature reserves (Table 6) and should be considered as a potential sources of water quality issues (Scanes et al. 2021).

Question four: Do sediment attributes selected for the NOF have an influence on biotic metrics?

Suren (2020) contends that the SAM2 protocol is not the best method to describe DFS in streams. This concern is partly based on the availability of alternative protocols. It also is grounded in the belief that the SAM2 method is particularly prone to inter-operator variability. The SAM2 protocol relies on visual assessment which can be subjective. Clapcott et al. (2011) reports a trial comparing SAM2 estimates of deposited sediment by 10 observers in three reaches with differing levels of bed sediment. The observers were able to accurately determine sites with low sediment levels; however, the variation in observations increased markedly as sediment levels increased. In the high sediment reach, some observers had difficulty agreeing that % sediment cover was greater than 50%, despite being supplied with diagrams to help estimates.

Clapcott et al. (2011) thus recommends training field staff to reduce variability and improve accuracy. If possible, taking photos will allow quality control of observations and enable digital image analysis (DIA) which has the potential to be a valuable tool (Turley et al. 2017), especially with the availability of increasingly powerful artificial intelligence (AI) software tools. In its defence (provided adequate training is implemented), the SAM2 method requires repeated measures (four observations from each of 20 plots on the streambed), helping to reduce the implicit biases in recordings made by different operators for a given site. Furthermore, in the context of the median values from five years of data collection for the NPS-FM, any inter-operator variability can be further mitigated. It should be noted that Clapcott et al. (2011) did not test the user-variability of the Shuffle method (SAM4) or the Wolman pebble count method (SAM3), which will also have biases.

Suren (2020) also correctly identifies a need for NOF attributes to have clearly demonstrable relationships with the biotic indices (e.g., MCI and %EPT). Clapcott et al. (2011) reports that the SAM2 protocol (% fine sediment) is correlated with the MCI (r = -0.38, p < 0.01, n = 168) and %EPT (r = -0.47, p < 0.01, n = 113), however, our results (Figure 11) suggest there may be exceptions to these general patterns with differences in climate, source of flow, geology, and other habitat attributes. Suren (2020) expresses reservations about using % sediment (based on the SAM3 protocol), because multiple regression models indicated that better relationships could be established between MCI and SAM4 sediment (Shuffle method) and sediment depth. A similar result was observed with %EPT.

However, it should be noted that sediment depth in the national protocols was not correlated with other measures of sediment and was poorly related to invertebrate biotic metrics (Clapcott et al. 2011). The poor performance of sediment depth with invertebrate metrics may reflect their underlying bias towards hard-bottomed streams (i.e., the MCI was developed using hard-bottomed streams in the Taranaki ring plain). Further, although the model selection procedure used by Suren

(2020) may have preferentially selected DFS predictors other than the SAM2 selected for the NOF, this does not mean that %DFS (SAM2) was wholly inadequate for showing a clearly demonstrable relationship with the biotic indices. Strong non-linear responses in invertebrate metrics (e.g., %EPT) to % fine sediment can mean that linear regression models are not optimal for describing the relationship between this environmental stressor and ecological endpoints (Burdon et al. 2013).

Question five: Will dynamic hydrogeomorphic processes associated with restoration affect the TAS?

In a final comment by Suren (2020), the author highlights a potentially vexing issue regarding dynamic hydrogeomorphic change following stream restoration involving riparian plantings. It should be noted that the same processes could play out in pasture converted to pine forestry (e.g., for carbon farming). Essentially, Suren (2020) describes that stream channels typically narrow with pastoral land-uses as grasses stabilise banks and enable greater bank storage of fine sediment. This process can be reversed with riparian plantings since the shading reduces groundcover and increases the potential for bank-stored sediment to be eroded and mobilised into the stream channel. Eventually the stream will reach a new equilibrium characterised by wider channels and lower sediment levels due to the stabilising influence of woody riparian vegetation (Table 6). This phenomenon is well described from research at Whatawhata in the Waikato (Davies-Colley 1997, Quinn et al. 1997). Consequently, the interim phase between these two states can result in considerable sediment mobilisation which may lead to restored and afforested streams breaching NBLs for DFS and other attributes. In an important study by Quinn et al. (1997), the authors found that %DFS was not significantly greater in recently planted pine streams, although it indicated the potential for this problem to manifest with greater suspendable DFS and smaller mean particle sizes in pine streams when compared to the pasture and native forest reference sites. However, another study showed that %DFS significantly decreased in a small pasture stream following native forest riparian planting (Quinn et al. 2009).

The potential problem of increased DFS might thus be determined by the type of plantings used and site-specific characteristics. The potential for post-planting sedimentation needs to be considered and contingencies planned for. This could take the form of commentary in BOPRC rules recognising that increases in DFS following riparian planting are expected. Furthermore, if it can be established that riparian planting has been done upstream of a site that falls below a NBL for DFS, then the increased DFS may reflect this NOP in the short term (Table 6), as banks readjust. Mitigation of these impacts could include best-practice planting (e.g., planting sedges and flaxes at channel margins, interplanting of enrichment species) and sediment traps to attenuate sediment transport and concentrate DFS in targeted areas (Hudson 2002, Guillozet et al. 2014).

Discussion of key points raised by Suren (2020)

A legitimate concern raised by Suren (2020) is that the REC classes outlined in Tables 24 and 25 of the NPS-FM (Appendix A in this document) are inappropriate for determining if a site is naturally soft-bottomed (SB) or is naturally hard-bottomed (HB) in the Bay of Plenty. Suren (2020) shows that using the REC classes means only one of the 149 sites currently monitored by the BOPRC as part of their SOE programme are SB. This is in stark contrast to the descriptions using the SAM3 protocol (Wolman method), which indicates that 80 are field-defined SB (> 50% of the streambed classified as sand or finer) with the remaining 69 sites characterised as HB. This discrepancy could require as many as 79 action plans for streams that are potentially naturally soft-bottomed, thus greatly increasing the council's current workload and delivering poor value to ratepayers by not prioritising the most urgent environmental needs.

However, Suren's (2020) proposed recommendation to use only the field data from the SAM3 protocol to determine if a site is naturally SB or HB is problematic. This method potentially underestimates DFS levels and using it alone relies on the *a priori* assumption that DFS is only the result of natural sources. This approach alone is undesirable as the council could be criticised for using an inappropriate method to avoid their statutory obligations. Ignoring present and persistent DFS problems in sites incorrectly designated as SB streams would expose the council to unmitigated risks. Here, we have provided a regression equation based on 25 streams in the Te Awanui / Tauranga Harbour catchment to help convert estimates of DFS using the SAM3 protocol (Wolman Pebble count) to their expected values using the SAM2 protocol prescribed in the NPS-FM (Figure 10).

Using the best available information (i.e., the most up-to-date and accurate geological maps) will provide critical information about the local geology influencing stream geomorphology. Our results indicate that hill source of flow streams in volcanic geologies can have elevated levels of DFS due to naturally occurring processes and yet still retain high ecological values. It would make sense for these sites to be granted an exemption for the DFS attribute, so that limited resources can be directed to streams that are under acute pressure from human activities. The information provided here could be used to help develop an alternative classification of stream type for the BOP region that is more appropriate than the REC classification provided in Tables 24 and 25 of the NPS-FM. In particular, the absence of an appropriate volcanic geology class in the REC database classifications to characterise areas with deep layers of tephra including fine ash and pumice should be rectified. Using a more nuanced approach that considers the complexities of the volcanic geology in the BOP along with topography and other factors related to source of flow will enable the BOPRC to better manage streams with naturally high levels of DFS.

Our analyses identified streams with hill source of flow and volcanic geology as potential REC classes to be granted an exception to the DFS attribute in the BOP region. However, another potential classification could also be based on the Freshwater Ecosystems of New Zealand (FENZ) geology and substrate size classification referred to in Figure 4 (see also Dare 2019). If these streams in the BOP region are all naturally SB, then the DFS bands in the NPS-FM are irrelevant. However, for all remaining HB streams, the DFS bands that are relevant remain to be determined. Existing NPS-FM DFS classes might be appropriate, otherwise an alternative scoring system could be developed (e.g., such as A: <10%; B: 10−20%; C: 20−30%; D: >30%). Any alternative scoring system would require further validation.

Looking to the future, a more definitive approach could also involve using additional data on sediment depth and loads, stream gradients and source of flow, along with the degree of hydrogeomorphic degradation (e.g., flow diversity; see Louhi et al. 2011, Hering et al. 2015), and the past and present intensity of land use in the upstream catchment. This integrated approach would provide a more objective way to determine the extent of naturally occurring processes resulting in DFS that does not meet bottom lines. Such an approach could apply the methods described by McDowell et al. (2013) to derive appropriate trigger values defined from the distribution of observed concentrations at pre-identified local reference sites. These trigger values can be defined by the 80th percentile of indicators considered harmful at high values (e.g., %DFS). Using sediment depth alone as an additional indicator of DFS could also be useful since it has previously been considered as a potential metric to assess naturally soft-bottomed streams in national protocol developments (Clapcott et al. 2011).

Our final recommendation is to assess if the best available information (e.g., the SAM3 protocol and/or other measures of DFS, up-to-date and accurate geological maps, supporting macroinvertebrate indicators) can be used to robustly determine if streams are naturally soft bottomed in the Bay of Plenty region. This work would further support the arguments made by the BOPRC that such exceptions to the REC classes are appropriate and well-justified.

Conclusions

Naturally occurring processes (NOP) impact the amount of deposited fine sediment (DFS) observed in many streams and rivers of the Bay of Plenty (BOP) region. We have identified the NOPs (e.g., geology, source of flow) that influence DFS levels in these streams and rivers (Table 6). Alongside these NOPs, we have indicated the likely increase in attribute state (i.e., to the B-D bands) depending on the mechanisms driving elevated DFS in these streams and rivers (Table 6). Our insights are important for BOPRC when considering appropriate target attribute states (TAS) and policies relating to improving DFS in the region. Alternative attribute state tables for DFS in the BOP need to consider the NOPs identified in Table 6 but will also benefit from additional data collection using the SAM2 protocol prescribed in the NPS-FM and conversion of existing SAM3 data. We have provided a regression equation based on 25 streams in the Te Awanui / Tauranga Harbour catchment to help convert estimates of DFS using the SAM3 protocol (Wolman Pebble count) to their expected values with the SAM2 protocol (Figure 10).

Summary

The NPS-FM (Clause 3.32) recognises that naturally occurring processes (NOP) may result in waterbodies not meeting NBLs. These exceptions mean the NPS-FM will allow councils to set TAS worse than a NBL (or Regional Bottom Line), but only if a TAS better than the bottom line is unable to be met due to NOP. Here we have reviewed evidence from the BOPRC and other sources about three key attributes set forth by the NPS-FM that have been identified as being impact by NOP in the Bay of Plenty. These attributes are:

- Dissolved Reactive Phosphorus (DRP)
- Lake Bottom Dissolved Oxygen (DO) and Mid-hypolimnetic DO
- Deposited Fine Sediment (DFS)

In our independent review we have evaluated the potential NOP in the Bay of Plenty Region and how these NPS-FM attributes (and associated bottom lines) may be inappropriate. We identified NOP for DRP (Table 1), Lake Bottom and Mid-Hypolimnetic DO (Table 4) and DFS (Table 6), and assessed what mechanisms underpin these processes, where they are likely to occur in the BOP and their potential impact on TAS.

In the case of DRP, natural sources of P associated with geothermal, volcanic and sedimentary geologies are driving high concentrations in BOP streams and rivers, although differences within the region rely less on geology and more on source of flow and climate. The widespread influence of catchment geology in the region results in attribute state bands for DRP due to NOP ranging from B-D bands depending on the processes and location (see Table 1). As potential alternative management guidelines, we have scaled the 80th percentile values from McDowell et al. (2013) to the median values for DRP concentrations in the BOP over the past five years to provide a ‘proof of concept’ for region-specific trigger values (Appendix D).

Regarding bottom water lake DO, lake trophic status, morphology, and thermal conditions are likely natural drivers of low DO in the Te Arawa lakes. The departure of the water quality in these lakes from reference conditions provided an indication of changes in bottom water DO related to anthropogenic drivers. This results in attribute state bands for lake-bottom and mid-hypolimnetic DO due to NOP ranging from B-D bands depending on the processes and characteristics in any given lake (see Table 4). Very preliminary and indicative TAS for DO attributes have been provided in Table 5 based on the discussion and analysis in this document.

The natural high levels of DFS in BOP streams and rivers are largely a consequence of the volcanic geology of the region, with deep layers of tephra including ash and fine pumice leading to streams dominated by fine substrates over large areas of the region. This results in attribute state bands for DFS due to NOP ranging from B-D bands depending on the processes and location (see Table 6). We presented evidence from the volcanic Kuratau catchment in the central North Island showing that naturally occurring fine sediment can lead to hill source of flow streams not meeting NBL for DFS despite having macroinvertebrate communities indicative of pristine conditions. Macroinvertebrate data from the BOP region shows that MCI scores in hill source of flow streams are largely invariant to the effects of catchment land use. Consequently, streams with REC classes of hill source of flow and volcanic geology could be granted an exception to the DFS attribute in the BOP region.

Our report has covered three NPS-FM attributes (DRP, bottom water lake DO, DFS) where the stated NOPs mean that the current and baseline states are below bottom lines, thus providing supporting evidence in accordance with clause 3.32(2) in the NPS-FM.

Literature cited

Abell, M. J., Dam-Bates, P., Özkundakci, D., Hamilton, D. P. (2020). Reference and current Trophic Level Index of New Zealand lakes: benchmarks to inform lake management and assessment. *New Zealand Journal of Marine and Freshwater Research* 54: 636-657

Allan, J. D., and Castillo, M. M. (2007). *Stream Ecology: Structure and Function of Running Waters*. 2nd edition. Springer, Dordrecht, The Netherlands.

Baldan, D., Kiesel, J., Hauer, C., Jähnig, S. C., & Hein, T. (2021). Increased sediment deposition triggered by climate change impacts freshwater pearl mussel habitats and metapopulations. *Journal of Applied Ecolog*y, 58, 1933– 1944

Beyá, J., Hamilton, D. P., and Burger, D. (2005). *Analysis of catchment hydrology and nutrient loads for Lakes Rotorua and Rotoiti.* CBER Report 74, University of Waikato, Hamilton, New Zealand.

Biggs, B. J. F. (2000). Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. *Journal of the North American Benthological Society* 19:17-31.

Bright, C. E., and Mager, S. M. (2016). Contribution of particulate organic matter to riverine suspended material in the Glendhu Experimental Catchments, Otago, New Zealand. *Journal of Hydrology* 55: 89-105.

Bruesewitz, D. A., Hamilton, D. P., and Shipper L. A. (2011). Denitrification potential in lake sediment increases across a gradient of catchment agriculture. *Ecosystems* 14: 341-352.

Bruns, N. M., Rutherford, J. C., and Clayton, J. S. (1999). A monitoring and classification system for New Zealand lakes and reservoirs. *Journal of Lake and Reservoir Management* 15(4): 255-271.

Burdon, F. J., Bai, Y., Reyes, M., Tamminen, M., Staudacher, P., Mangold, S., Singer, H., Räsänen, K., Joss, A., Tiegs, S. D., Jokela, J., Eggen, R. I. L., and Stamm, C. (2020). Stream microbial communities and ecosystem functioning show complex responses to multiple stressors in wastewater. *Global Change Biology* 26: 6363– 6382

Burdon, F. J., McIntosh, A. R., and Harding, J. S. (2013). Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. *Ecological Applications* 23: 1036-1047

Charlton, M. N. (1980). Hypolimnion consumption of lakes: discussion of productivity and morphometry effects. *Canadian Journal of Fisheries and Aquatic Sciences* 75: 1531-1539.

Clapcott, J. E., Young, R. G., Harding, J. S., Matthaei, C. D., Quinn, J. M., and Death, R. G. (2011). *Sediment Assessment Methods: protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values*. Cawthron Institute, Nelson, New Zealand.

Collier, K. J., and Bowman, E. J. (2003). Role of wood in pumice-bed streams: I: Impacts of post-harvest management on water quality, habitat and benthic invertebrates. *Forest Ecology and Management* 177: 243-259

Collier, K. J., and Halliday, J. N. (2000). Macroinvertebrate-wood associations during decay of plantation pine in New Zealand pumice-bed streams: stable habitat or trophic subsidy? *Journal of the North American Benthological Society* 19:94-111

Collier, K. J., and Smith, B. J. (2003). Role of wood in pumice-bed streams: II: Breakdown and colonisation. *Forest Ecology and Management* 177:261-276

Cornett, R.J., and Rigler, F. H. (1987). Decomposition of seston in the hypolimnion. *Canadian Journal of Fisheries and Aquatic Sciences* 44: 146-151.

Dare, J. (2019). *Confidential Proposed NPS-FM Attribute Tables for Nutrients*. Internal Memorandum to Nicola Green, Principal Advisor, Policy and Planning dated 8 August, 2019. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Davies-Colley, R. J. (1997). Stream channels are narrower in pasture than in forest. *New Zealand Journal of Marine and Freshwater Research* 31: 599-608

Davis, N. G., Hodson, R., and Matthaei, C. D. (2022). Long-term variability in deposited fine sediment and macroinvertebrate communities across different land-use intensities in a regional set of New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* 56:191-212.

Deeds, J., Amirbahman, A., Norton, S. A., Suitor, D. G., Bacon, L. C. (2021). Predicting anoxia in low-nutrient temperate lakes. *Ecological Applications* 31: e02361.

Delany, A. (2022). *Modeled organic carbon, dissolved oxygen, and Secchi for six Wisconsin Lakes, 1995-2014* [Data set]. Environmental Data Initiative.

Dillon, P. J., and Kirchner, W. B. (1975). The effects of geology and land use on the export of phosphorus from watersheds. *Water Research* 9: 135-148

Dingle, E. H., Sinclair, H. D., Venditti, J. G., Attal, M., Kinnaird, T. C., Creed, M., Quick, L., Nittrouer, J. A., and Gautam, D. (2020). Sediment dynamics across gravel-sand transitions: Implications for river stability and floodplain recycling. *Geology* 48:468-472

Dordoni, M., Seewald, M., Rinke, K., van Geldern, R., Schmidmeier, J., & Barth, J. A. C. (2022). Mineralization of autochthonous particulate organic carbon is a fast channel of organic matter turnover in Germany’s largest drinking water reservoir. *Biogeochemistry* 19: 5343-5355

Dymond, J. R., Ausseil, A. G., Parfitt, R. L., Herzig, A., and McDowell, R. W. (2013). Nitrate and phosphorus leaching in New Zealand: a national perspective. *New Zealand Journal of Agricultural Research* 56: 49-59

Guillozet, P., Smith, K., and Guillozet, K. (2014). The rapid riparian revegetation approach. *Ecological Restoration* 32: 113-124

Hamill, K. (2022). *Trophic Level Index review of targets and variability for Rotorua Lakes.* Prepared for Bay of Plenty Regional Council. Whakatāne, New Zealand.

Healey, B. (2005). *Minerals in the Bay of Plenty Region*. Resource Policy Publication 2005/01, Environment Bay of Plenty (BOPRC), Whakatāne, New Zealand.

Hering, D., Aroviita, J., Baattrup-Pedersen, A., Brabec, K., Buijse, T., Ecke, F., Friberg, N., Gielczewski, M., Januschke, K., Köhler, J., Kupilas, B., Lorenz, A. W., Muhar, S., Paillex, A., Poppe, M., Schmidt, T., Schmutz, S., Vermaat, J., Verdonschot, P. F. M., Verdonschot, R. C. M., Wolter, C., and Kail, J. (2015). Contrasting the roles of section length and instream habitat enhancement for river restoration success: a field study of 20 European restoration projects. *Journal of Applied Ecology* 52:1518-1527

Hoellein, T.J., Bruesewitz, D.A., Hamilton, D.P. (2012) Are geothermal streams important sites of nutrient uptake in an agricultural and urbanising landscape (Rotorua, New Zealand)? *Freshwater Biology* 57: 116-128

Hudson, H. R. (2002). *Development of an in-channel coarse sediment trap best management practice.* Environmental Management Associates Ltd Report 2002-10, Ministry of Agriculture and Forestry, Wellington, New Zealand.

Hume, T., Green, M., and Elliott, S. (2010). *Tauranga Harbour sediment study: assessment of predictions for management.* Niwa Institute of Water & Atmospheric Research Ltd

Jackson, M. C., Loewen, C. J. G., Vinebrooke, R. D., and Chimimba, C. T. (2016). Net effects of multiple stressors in freshwater ecosystems: a meta-analysis. *Global Change Biology* 22:180-189

Jones, J. I., Murphy, J. F., Collins, A. L., Sear, D. A., Naden, P. S., and Armitage, P. D. (2012). The impact of fine sediment on macro-invertebrates. *River Research and Applications* 28:1055-1071

Johnson, C. J. (2013). Identifying ecological thresholds for regulating human activity: Effective conservation or wishful thinking? *Biological Conservation* 168: 57-65

Jolly, V. H. (1967). The comparative limnology of some New Zealand lakes: 1. Physical and chemical. *New Zealand Journal of Marine and Freshwater Research* 2: 214-259.

Kilroy, C., Snelder, T., and Stoffels, R. J. (2020). *Periphyton-environment relationships in the Bay of Plenty: Analysis of data from 2015-2019*. NIWA client report 2020179CH.

Konrad, C. and Gellis, A. (2018), Factors Influencing Fine Sediment on Stream Beds in the Midwestern United States. *Journal of Environmental Quality*, 47: 1214-1222

Kpodonu, A. T. N. K. (2016). *Temporal variability in the water quality of a deep temperate oligotrophic lake* (Thesis, Doctor of Philosophy). University of Waikato, Hamilton, New Zealand.

Larned, S. T., Scarsbrook, M. R., Snelder, T. H., and Norton, N. (2005). *Nationwide and regional state and trends in river water quality 1996-2002*. Report for the Ministry for the Environment, NIWA Client Report: CHC2003-051, National Institute of Water and Atmospheric Research, Christchurch, New Zealand.

Leathwick, J. R., Snelder, T., Chadderton, W. L., Elith, J., Julian, K., and Ferrier, S. (2011). Use of generalised dissimilarity modelling to improve the biological discrimination of river and stream classifications. *Freshwater Biology* 56:21-38

Louhi, P., Mykrä, H., Paavola, R., Huusko, A., Vehanen, T., Mäki-Petäys, A., and Muotka, T. (2011). Twenty years of stream restoration in Finland: little response by benthic macroinvertebrate communities. *Ecological Applications* 21:1950-1961

Lowe, D. J. (1990). Tephra studies in New Zealand: an historical review. *Journal of the Royal Society of New Zealand* 20:119-150

Mainstone, C. P., and Parr, W. (2002). Phosphorus in rivers — ecology and management. *Science of The Total Environment* 282-283:25-47

Matthaei, C. D., Piggott, J. J., and Townsend, C. R. (2010). Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. *Journal of Applied Ecology* 47:639-649

McBride, C. G., Mac Cormick, A., and Verburg, P. (2020). *Estimated catchment loads of nitrogen and phosphorus to the Rotorua Te Arawa Lakes – Catchment, atmospheric and geothermal inputs*. ERI report 143 prepared for the Bay of Plenty Regional Council.

McColl, R. (1972). Chemistry and trophic status of seven New Zealand lakes. New Zealand Journal of *Marine and Freshwater Research* 6:399-447

McDowell RW, Biggs BJF, Sharpley AN, Nguyen L (2004) Connecting phosphorus loss from land to surface water quality. *Chemistry and Ecology* 20:1-40

McDowell, R. W., Larned, S. T., and Houlbrooke, D. J. (2009). Nitrogen and phosphorus in New Zealand streams and rivers: Control and impact of eutrophication and the influence of land management. *New Zealand Journal of Marine and Freshwater Research* 43:985-995

McDowell, R. W. (2010). The efficacy of strategies to mitigate the loss of phosphorus from pastoral land use in the catchment of Lake Rotorua. AgResearch report for the Bay of Plenty Regional Council. 32 p.

McDowell, R. W., Snelder, T., and Cox, N. (2013*). Establishment of reference conditions and trigger values for chemical, physical and micro-biological indicators in New Zealand streams and rivers.* AgResearch, Mosgiel, New Zealand.

McEwan, W. M., (1987). *Ecological regions and districts of New Zealand. Booklet to accompany sheet 2: descriptions of Districts in the central North Island, from Meremere to Eastern Hawkes Bay.* Department of Conservation, Wellington, New Zealand.

McIntosh, J. (2012). *Alum dosing of two stream discharges to Lake Rotorua*. Internal report prepared February 2012. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Meybeck, M. (1982). Carbon, nitrogen, and phosphorus transport by world rivers. *American Journal of Science* 282:401-450

Meyers, P. A. (1994). Preservation of elemental and isotopic source identification of sedimentary organic matter. *Chemical Geology* 114: 289–302.

Moore, T. R., 1989. Dynamics of dissolved organic carbon in forested and disturbed catchments, Westland, New Zealand. *Water Resources Research* 25: 1321-1330.

Morgan, P. G. (1919). *The limestone and phosphate resources of New Zealand: considered principally in relation to agriculture.* Dept. of Mines, Geological Survey Branch, Wellington, New Zealand.

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

Mortimer, N., and Strong, D. T. (2014). New Zealand limestone purity. *New Zealand Journal of Geology and Geophysics* 57:209-218

Müller, B., Bryant, L. D., Matzinger, A., Wüst, A. (2012). Hypolimnetic oxygen depletion in eutrophic lakes. *Environmental Science and Technology* 46:9964-9971.

Munn, N. L., and Meyer, J. L. (1990). Habitat-specific solute retention in two small streams: an intersite comparison. *Ecology* 71:2069-2082

Neverman, A. J., Donovan, M., Smith, H. G., Ausseil, A.-G., and Zammit, C. (2023). Climate change impacts on erosion and suspended sediment loads in New Zealand. *Geomorphology* 427:108607

Newsome, P. F. J. (1992). *New Zealand Land Resource Inventory: ARC/INFO Data Manual: Edition 1, May 1992*. DSIR Land Resources, Department of Scientific and Industrial Research.

Nürnberg, G. K. (2004). Quantified hypoxia and anoxia in lakes and reservoirs. *Scientific World Journal* 4:42-54

Parshotam, A., Hume, T., Elliott, S., Green, M., and Wadhwa, S. (2008). *Tauranga Harbour Sediment Study: Specification of Scenarios.* Client Report HAM2008-117 prepared for the Bay of Plenty Regional Council. National Institute of Water & Atmospheric Research Ltd., Hamilton, New Zealand.

Pingram, M. A., Collier, K. J., Özkundakci, D., and Garrett-Walker, J. (2020). Food web characteristics of fish communities across degraded lakes provide insights for management in multi-stressor environments. *Aquatic Ecology* 54: 401-419

Porder, S., and Ramachandran, S. (2013). The phosphorus concentration of common rocks—a potential driver of ecosystem P status. *Plant and Soil* 367:41-55

Pringle, C. M., Rowe, G. L., Triska, F. J., Fernandez, J. F., and West, J. (1993). Landscape linkages between geothermal activity and solute composition and ecological response in surface waters draining the Atlantic slope of Costa Rica. *Limnology and Oceanography* 38:753-774

Quinn, J. M., Cooper, A. B., Davies-Colley, R. J., Rutherford, J. C., and Williamson, R. B. (1997). Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research* 31:579-597

Quinn, J. M., Croker, G. F., Smith, B. J., and Bellingham, M. A. (2009). Integrated catchment management effects on flow, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research* 43:775-802

Quinn, J. M., and Vickers, M. L. (1992). *Benthic invertebrates and related habitat factors in the Tongariro River*. Water Quality Centre Consultancy Report No. 6025/2. DSIR Water Quality Centre, Hamilton, New Zealand.

Ramos, M. C., Lizaga, I., Gaspar, L., and Navas, A. (2022). The impacts of exceptional rainfall on phosphorus mobilisation in a mountain agroforestry catchment (NE, Spain). *CATENA* 216: 106407.

Rast, W., and Lee, G. F. (1978). *Summary analysis of the North American OECD Eutrophication Project: Nutrient loading-lake response relationship and trophic status indices*. U.S. EPA Report No. EPA/3-78-008. Ecological Research Series, U.S. Environmental Protection Agency, Corvallis, USA.

Rijkse, W. C. and Guinto, D. F. (2010). *Soils of the Bay of Plenty Volume 1* Western Bay of Plenty Environmental Publication 2010/11-1. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Saeed, H. (2016). *Redox cycling of colloidal macro- and micro-nutrients in a monomictic lake* (Thesis, Doctor of Philosophy). University of Waikato, Hamilton, New Zealand.

Sand-Jensen, K. (1998). Influence of submerged macrophytes on sediment composition and near-bed flow in lowland streams. *Freshwater Biology* 39:663-679

Santoso, A. B., Hamilton, D. P., Hendy, C. H., and Shipper, L.A. (2017). Carbon dioxide emissions and sediment organic carbon burials across a gradient of trophic state in eleven New Zealand lakes. *Hydrobiologia* 795: 341-354.

Scanes, P. R., McSorley, A., and Dickson, A. (2021). Feral horses (*Equus caballus*) increase suspended sediment in subalpine streams. *Marine and Freshwater Research* 72:1290-1302

Scarsbrook, M.R., McBride, C.G., McBride, G.B. and Bryers, G.G. (2003). Effects of Climate Variability on Rivers: Consequences for Long Term Water Quality Analysis. *Journal of the American Water Resources Association* 39: 1435-1447.

Schallenberg, M., Burns, C. W. (1999). Does zooplankton grazing affect seston size structure and areal hypolimnetic oxygen depletion in lakes? *Archiv für Hydrobiologie* 147: 1-24.

Scholes, P. (2021). *Advice on NPS-FW Attributes Mid and bottom Dissolved Oxygen and Dissolved Reactive Phosphorus.* Internal Memorandum to Rochelle Carter and Nicola Green, Principal Advisors Science, and Policy and Planning dated 3 February 2021. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Scholes, P., and Hamill, K. (2016). *Rotorua Lakes water quality report 2014/2015.* Environmental Publication 2016/06. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Snelder, T. H., and Biggs, B. J. F. (2002). Multi-scale river environment classification for water resources management. *Journal of the American Water Resources Association* 38:1225-1239

Snelder, T. H., Cattanéo, F., Suren, A. M., and Biggs, B. J. F. (2004). Is the River Environment Classification an improved landscape-scale classification of rivers? *Journal of the North American Benthological Society* 23: 580-598

Snelder, T. H., Dey, K., and Suren, A. M. (2019). *Analysis of habitat factors influencing invertebrate communities in streams of the Bay of Plenty Region.* Client report prepared for the Bay of Plenty Regional Council. LWP Ltd., Lyttelton, New Zealand.

Suren, A. (2020). *Commentary of NOF Deposited Sediment Attributes*. Internal Memorandum to Rochelle Carter and Nicola Green, Principal Advisors Science, and Policy and Planning dated 25 August 2020. Bay of Plenty Regional Council, Whakatāne, New Zealand.

Tempero, G., McBride, C., Abell, J., and Hamilton, D. (2015). *Anthropogenic phosphorus loads to Lake Rotorua.* Client report prepared for the Bay of Plenty Regional Council. Environmental Research Institute Report No. 66. The University of Waikato, Hamilton. 31 p.

Thomas, G. W., and Crutchfield, J. D. (1974). Nitrate-nitrogen and phosphorus contents of streams draining small agricultural watersheds in Kentucky. *Journal of Environmental Quality* 3:46-49

Timmins, S. M. (1983). Mt Tarawera: 1. Vegetation types and successional trends. *New Zealand Journal of Ecology* 6: 99-105.

Triska, F. J., Pringle, C. M., Duff, J. H., Avanzino, R. J., Ramirez, A., Ardon, M., and Jackman, A. P. (2006). Soluble reactive phosphorus transport and retention in tropical, rainforest streams draining a volcanic and geothermally active landscape in Costa Rica: Long-term concentration patterns, pore water environment and response to ENSO events. *Biogeochemistry* 81:131-143

Trolle, D., Hamilton, D. P., Hendy, C., and Pilditch, C. (2008). Sediment and nutrient accumulation rates in sediments of twelve New Zealand lakes: Influence of lake morphology, catchment characteristics and trophic state. *Marine and Freshwater Research* 59: 1067–1078.

Turley, M.D., Bilotta, G.S., Arbociute, G., Chadd, R.P., Extence, C.A., Brazier, R.E. (2017) Quantifying submerged deposited fine sediments in rivers and streams using digital image analysis. *River Research and Applications* 33: 1585-1595

Vale, S., Smith, H., Neverman, A., and Herzig, A. (2021). *Application of SedNetNZ with erosion mitigation and climate change scenarios and temporal disaggregation in the Bay of Plenty region.* Contract Report LC4002 prepared for the Bay of Plenty Regional Council. Manaaki Whenua – Landcare Research, New Zealand.

Wagenhoff, A., Townsend, C. R., and Matthaei, C. D. (2012). Macroinvertebrate responses along broad stressor gradients of deposited fine sediment and dissolved nutrients: a stream mesocosm experiment. *Journal of Applied Ecology* 49:892-902

Wallace J.B., Benke A.C. (1984) Quantification of wood habitat in subtropical coastal plain streams. *Canadian Journal of Fisheries and Aquatic Sciences* 41:1643-1652.

Wentworth, C. K. (1922). A Scale of Grade and Class Terms for Clastic Sediments. *The Journal of Geology* 30: 377-392

White, E. (1972). The distribution and movement of 'reactive' phosphorus through catchments under varied land use. *Proceedings of the New Zealand Ecological Society* 19:163-172

Wilkinson, G. M., Pace, M. L., and Cole, J. J. (2013). Terrestrial dominance of organic matter in north temperate lakes: organic matter composition in lakes. *Global Biogeochemical Cycles* 27: 43-51.

Wills, T. J., Retallick, R. W. R., Greet, J., and Bennett, A. (2023). Browsing by non-native invasive sambar deer dramatically impacts forest structure. *Forest Ecology and Management* 543:121153.

Wilson, N. J. (2022). *Flows of water and nutrients to Lake Tarawera and connected lakes.* (Thesis, Master of Science). University of Waikato, Hamilton, New Zealand.

Wolman, M. G. (1954). A method of sampling coarse river-bed material. *Eos - Transactions American Geophysical Union* 35:951-956

Young, R. G., and Huryn, A. D. (1999). Effects of land use on stream metabolism and organic matter turnover. *Ecological Applications* 9: 1359-1376.

Appendix A – Additional information from the NPS-FM 2020

Appendix 2C – Sediment classification tables

In this Appendix, **REC groups** refers to the classes and categories described in the New Zealand River Environment Classification User Guide (*see* clause 1.8), except where those REC groups are further clustered according to table 26.

Table 23 Suspended sediment class composition

Suspended sediment class	Suspended sediment clustered River Environment Classification groups
1	CD_Low_HS; WW_Low_VA; WW_Hill_VA; CD_Low_Al; CW_Hill_SS; CW_Mount_SS; CW_Hill_VA; CD_Hill_SS; CD_Hill_VA; CD_Low_VA; CW_Low_VA; CW_Mount_VA; CW_Mount_HS; CD_Mount_Al; CW_Hill_Al; CW_Mount_Al; WD_Low_Al
2	CD_Low_SS; WW_Low_HS; WW_Low_SS; WW_Hill_HS; WW_Hill_SS; WW_Low_Al; WD_Low_SS; WD_Lake_Any; WD_Low_HS; WD_Low_VA
3	CW_Hill_HS; CW_Lake_Any; CD_Lake_Any; WW_Lake_Any; CW_Low_HS; CW_Low_Al; CD_Hill_HS; CD_Hill_Al; CD_Mount_HS; CD_Mount_SS; CD_Mount_VA
4	CW_Low_SS

Table 24 – Deposited sediment class composition

Deposited sediment class	Deposited sediment clustered River Environment Classification groups
1	WD_Low_HS; WW_Lake_Any
2	CD_Hill_Al; CD_Low_HS; CD_Low_VA; WW_Low_HS; WW_Low_VA; CD_Hill_SS; CD_Lake_Any; CW_Lake_Any; CW_Low_Al; CD_Hill_HS; CW_Hill_VA; CW_Low_SS; CW_Low_VA
3	CD_Low_Al; CD_Low_SS; WW_Hill_SS; WW_Low_SS
4	CD_Hill_VA; CW_Mount_VA; WW_Hill_HS; CW_Mount_SS; CD_Mount_Al; CD_Mount_HS; CD_Mount_SS; CD_Mount_VA; CW_Hill_Al; CW_Hill_HS; CW_Hill_SS; CW_Low_HS; CW_Mount_Al; CW_Mount_HS; WW_Hill_VA

Table 25 – Clustered River Environment Classification groups that are naturally soft-bottomed

WD_Low_Al; WD_Low_VA; WD_Lake_Any; WD_Low_SS; WW_Low_Al

Table 26 – Further clustering of River Environment Classification groups specific to this appendix

REC variable	REC groups	Clustered REC groups
Climate	Warm-Wet	Warm-Wet (WW)
	Warm-Extremely Wet	
	Warm-Dry	Warm-Dry (WD)
	Cold-Wet	Cold-Wet (CW)
	Cold-Extremely Wet	
	Cold-Dry	Cold-Dry (CD)
Topography (Source of flow)	Lowland	Lowland (Low)
	Lakefed	Lakefed (Lake)
	Hill	Hill (Hill)
	Mountain	Mountain (Mount)
	Glacial Mountain	
Geology	Soft Sedimentary	Soft Sedimentary (SS)
	Plutonic Volcanic	
	Miscellaneous	
	Hard Sedimentary	Hard Sedimentary (HS)
	Alluvium	Alluvium (Al)
	Volcanic Basic	Volcanic (VA)
	Volcanic Acidic	

Appendix B – Additional results from LAWA data

Table B1. Mean and median (± 95% confidence intervals) catchment land cover (%) in New Zealand streams and rivers. Sites have been grouped into North and South Islands to compare with the Bay of Plenty Region. The North Island sites excluding the Bay of Plenty include the Auckland, Gisborne, Hawkes Bay, Wellington, Manawatū-Whanganui, Northland, Taranaki, and Waikato regions. Land cover indicates broad categories assigned in the River Environment Classification. Raw data for 2018 sourced from the LAWA website (www.lawa.org.nz).

Group	Land cover	Sites	Mean	Lower CI	Upper CI	Median	Lower CI	Upper CI
Bay of Plenty	Native veg.	55	46.6	38.0	55.1	43.0	21.5	59.1
	Exotic forest	51	15.6	10.9	20.3	8.6	0.1	12.7
	Pasture	52	36.0	29.0	42.9	32.2	21.2	40.4
	Urban	33	5.4	2.0	8.9	1.1	-3.3	1.9
North Island	Native veg.	236	37.2	33.3	41.0	30.7	25.3	35.2
	Exotic forest	212	11.4	9.3	13.4	6.3	4.9	7.7
	Pasture	217	46.4	42.8	50.0	47.6	43.6	55.0
	Urban	161	13.5	9.2	17.7	0.8	0.4	1.1
South Island	Native veg.	168	41.2	36.7	45.7	37.8	29.5	45.9
	Exotic forest	143	7.9	6.1	9.7	3.6	1.8	4.4
	Pasture	158	43.4	38.7	48.2	42.8	35.2	52.7
	Urban	108	6.3	3.3	9.3	0.3	0.0	0.4

Table B2. Mean and median (± 95% confidence intervals) Macroinvertebrate Community Index (MCI) scores in New Zealand streams and rivers. Sites have been grouped into North and South Islands to compare with the Bay of Plenty Region. The North Island sites excluding the Bay of Plenty include the Auckland, Gisborne, Hawkes Bay, Wellington, Manawatū-Whanganui, Northland, Taranaki, and Waikato regions. Land cover indicates broad categories assigned in the River Environment Classification database. Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz).

Group	Land cover	Sites	Mean	Lower CI	Upper CI	Median	Lower CI	Upper CI
Bay of Plenty	Native veg.	48	114	110	118	115	110	118
	Exotic forest	23	112	107	118	115	111	122
	Pasture	56	103	99	108	104	96	110
	Urban	4	95	74	115	89	65	93
North Island	Native veg.	121	123	121	125	124	121	127
	Exotic forest	20	103	97	109	102	95	110
	Pasture	361	94	92	96	97	96	100
	Urban	36	76	70	82	72	61	79
South Island	Native veg.	154	110	108	111	110	107	113
	Exotic forest	10	109	99	118	112	106	116
	Pasture	240	95	94	97	96	95	98
	Urban	20	79	73	85	72	60	75

Table B3. Analysis of Variance (ANOVA) results from the global mixed-model testing additive Macroinvertebrate Community Index (MCI) responses in Bay of Plenty streams to different River Environment Classification (REC) categories (climate, geology, source of flow, and land cover). The raw data for the past five years (2018-2022) used in the mixed model was sourced from the LAWA website (www.lawa.org.nz).

Predictor	*F*-value	*df*	*df* residuals	*P*-value
Climate	2.300	3	100.884	0.082
Geology	4.548	3	98.593	**0.005**
Source of Flow	4.936	2	115.816	**0.009**
Land cover	4.940	3	86.027	**0.003**

Table B4. Post-hoc results from a mixed-model testing additive Macroinvertebrate Community Index (MCI) responses in Bay of Plenty streams to different River Environment Classification (REC) land cover categories. Land cover indicates broad categories assigned in the REC database (native vegetation, exotic forestry, pasture, and urban). The climate predictor was dropped from this model (ΔAIC = -11.1). The raw data for the past five years (2018-2022) used in the mixed model was sourced from the LAWA website (www.lawa.org.nz).

Contrast	Estimate	SE	*df*	*t*-ratio	*P*-value
Native - Exotic forest	-0.008	0.038	93.894	-0.208	0.997
Native - Pasture	0.085	0.030	78.064	2.852	**0.028**
Native - Urban	0.191	0.071	116.337	2.689	**0.040**
Exotic forest - Pasture	0.093	0.039	84.043	2.380	0.089
Exotic forest - Urban	0.199	0.076	110.935	2.621	**0.048**
Pasture - Urban	0.105	0.070	116.267	1.509	0.436

Table B5. Post-hoc results from a mixed-model testing additive Macroinvertebrate Community Index (MCI) responses in Bay of Plenty streams to different River Environment Classification (REC) source of flow categories. Source of flow indicates broad categories assigned in the REC database (hill, lowland, and lake-outlet). The climate predictor was dropped from this model (ΔAIC = -11.1). The raw data for the past five years (2018-2022) used in the mixed model was sourced from the LAWA website (www.lawa.org.nz).

Contrast	Estimate	SE	*df*	*t*-ratio	*P*-value
Hill - Lowland	0.098	0.031	88.732	3.126	**0.007**
Hill - Lake	0.214	0.074	121.493	2.892	**0.013**
Lowland - Lake	0.116	0.070	121.576	1.654	0.227

Table B6. Post-hoc results from a mixed-model testing Macroinvertebrate Community Index (MCI) responses in Bay of Plenty streams to different River Environment Classification (REC) geology categories. Geology indicates the broad categories assigned in the REC database (native vegetation, exotic forestry, pasture, and urban). The climate predictor was dropped from this model (ΔAIC = -11.1). The raw data for the past five years (2018-2022) used in the mixed model was sourced from the LAWA website (www.lawa.org.nz).

Contrast	Estimate	SE	*df*	*t*-ratio	*P*-value
Alluvium – Hard Sedimentary	-0.500	0.146	136.769	-3.427	**0.004**
Alluvium – Soft Sedimentary	-0.581	0.168	132.161	-3.465	**0.004**
Alluvium – Volcanic	-0.495	0.140	140.199	-3.536	**0.003**
Hard Sedimentary – Soft Sedimentary	-0.081	0.101	111.975	-0.801	0.854
Hard Sedimentary – Volcanic	0.004	0.041	54.823	0.101	1.000
Soft Sedimentary – Volcanic	0.085	0.095	113.534	0.900	0.805

Table B7. Analysis of Variance (ANOVA) results from a mixed-model testing interactive Macroinvertebrate Community Index (MCI) responses in Bay of Plenty streams to two River Environment Classification (REC) categories (source of flow and land cover). This model had a lower AIC score than the global additive model, indicating a better fit (ΔAIC = -13.9). The raw data for the past five years (2018-2022) used in the mixed-model was sourced from the LAWA website (www.lawa.org.nz).

Predictor	*F*-value	*df*	*df* residuals	*P*-value
(Intercept)	18484.726	1	104.073	**<0.001**
Source of Flow	6.044	2	115.182	**0.003**
Land cover	0.738	3	117.688	0.531
Source : Land cover	4.502	5	115.379	**0.001**

Table B8. Post-hoc results from a mixed-model testing MCI responses in BOP streams with different sources of flow to catchment land cover. Land cover indicates broad categories assigned in the REC database (native vegetation, exotic forestry, pasture, and urban). The raw data for the past five years (2018-2022) used in the mixed-model was sourced from the LAWA website (www.lawa.org.nz).

Source of Flow	Contrast	Estimate	SE	*df*	*t*-ratio	*P*-value
Hill	Native - Exotic forest	0.077	0.100	113.751	0.771	0.867
	Native - Pasture	0.078	0.056	111.132	1.387	0.510
	Native - Urban	0.086	0.141	133.179	0.613	0.928
	Exotic forest - Pasture	0.001	0.102	112.588	0.014	1.000
	Exotic forest - Urban	0.009	0.167	129.661	0.056	1.000
	Pasture - Urban	0.008	0.139	119.460	0.057	1.000
Lowland	Native - Exotic forest	-0.029	0.039	102.758	-0.725	0.887
	Native - Pasture	0.122	0.031	106.391	3.914	**0.001**
	Native - Urban	0.235	0.079	99.993	2.978	**0.019**
	Exotic forest - Pasture	0.151	0.037	96.710	4.027	**0.001**
	Exotic forest - Urban	0.263	0.081	101.200	3.247	**0.008**
	Pasture - Urban	0.112	0.078	96.565	1.442	0.477
Lake	Native - Exotic forest	0.111	0.178	95.839	0.624	0.924
	Native - Pasture	-0.417	0.162	118.900	-2.569	0.055
	Native - Urban					
	Exotic forest - Pasture	-0.528	0.160	112.898	-3.295	**0.007**
	Exotic forest - Urban					
	Pasture - Urban					

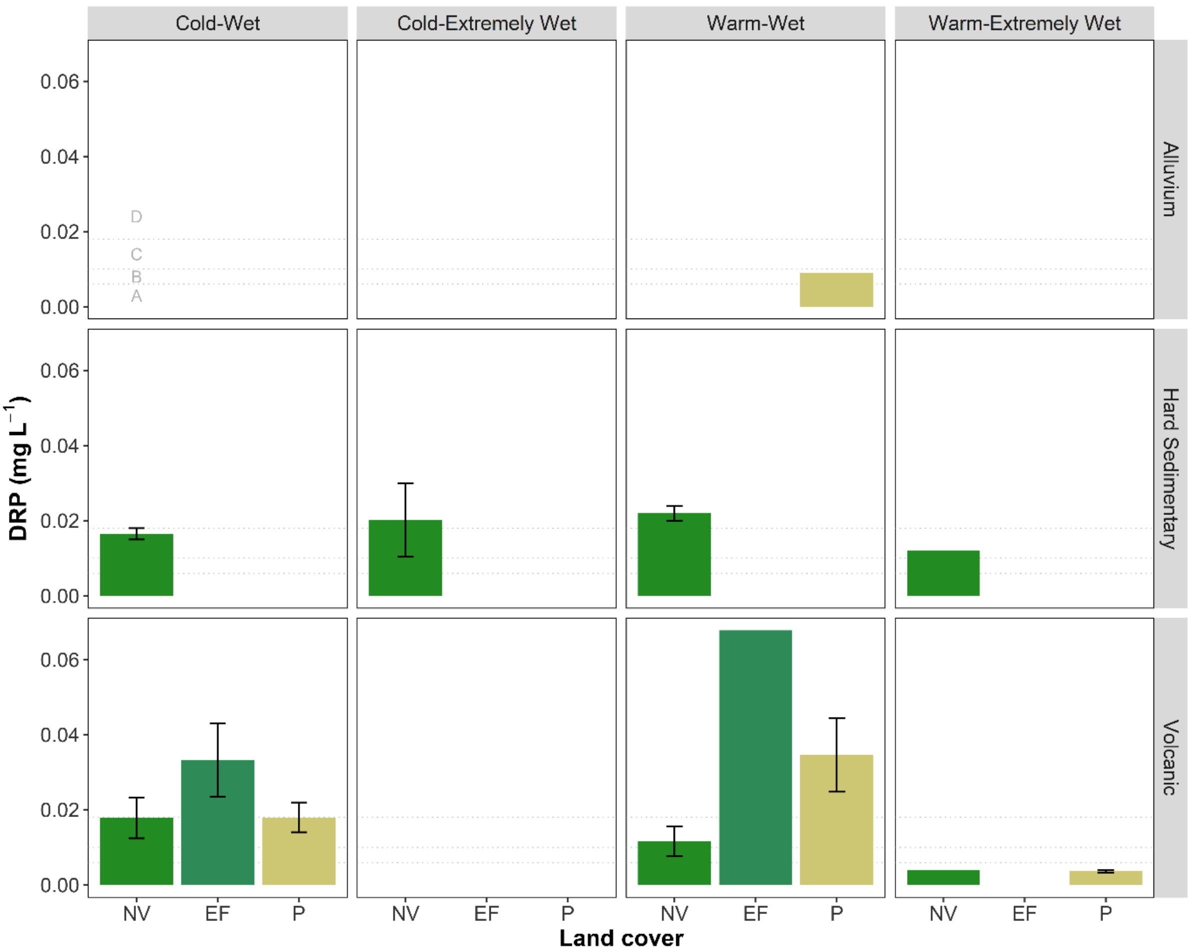

Figure B1. Mean concentrations (± 1 S.E.) of dissolved reactive phosphorus (mg/L) in Bay of Plenty streams and rivers. Sites have been grouped into River Environment Classification (REC) categories for climate and geology. Land cover indicates broad categories assigned in the REC (NV, Native vegetation; EF, Exotic forestry; P, Pasture; U, Urban). Raw data for the past five years (2018-2022) sourced from the LAWA website (www.lawa.org.nz).

Appendix C – Additional results showing NOP that affect DRP

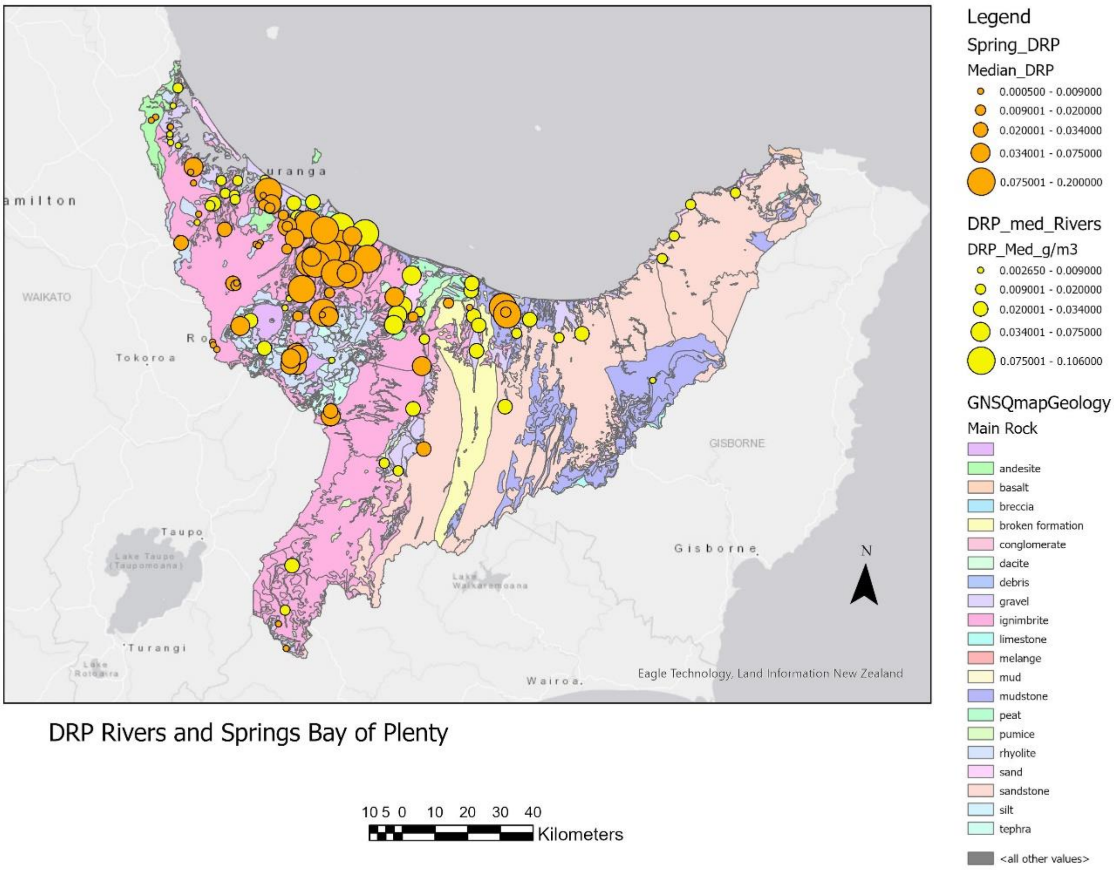

Figure C1. Monitoring data made available by Paul Scholes (BOPRC) highlighting the elevated dissolved reactive phosphorus (DRP) concentrations in groundwater-fed springs of the Bay of Plenty Region when compared to rivers. The concentrations are expressed as g/m^3 which is equivalent to mg/L.

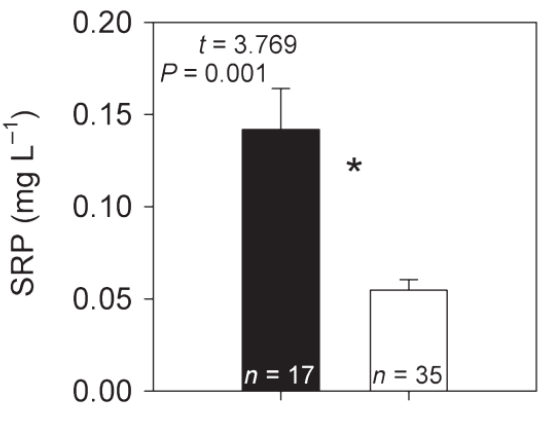

Figure C2. Results from Hoellein et al. (2012) showing differences in soluble reactive phosphorus (SRP) concentrations between geothermal and non-geothermal streams in the Bay of Plenty Regional Council (BOPRC) survey of streams in the Rotorua area. SRP is the same attribute as dissolved reactive phosphorus (DRP). Significant difference (t-test) shown with asterisk. Error bars were unstated in the original publication but are likely standard errors.

Appendix D – Potential trigger values from McDowell et al. (2013)

Table D1. Observed median and estimated trigger values for DRP concentrations (mg/L) in the Bay of Plenty based on McDowell et al. (2013). Those authors estimated median and 80th percentiles for DRP concentrations (data permitting) along with the 95% confidence interval (CI) of the estimates at the 3rd level (climate - C by topography/source of flow - SOF by geology - G) of the REC in New Zealand. We have scaled the trigger values (80%ile) of McDowell et al. (2013) to the median DRP concentrations at each sampling location in the Bay of Plenty (sorted by REC land cover type). This approach provides *potential* trigger values for a BOP region-specific exemption to the NPS-FM.

Land cover	Site name	C	SOF	G	Bay of Plenty				McDowell et al. (2013)					
					Median	Trigger	Lower CI	Upper CI	Median	Lower CI	Upper CI	80%ile	Lower CI	Upper CI
Native	Haparapara at SH35	CW	H	HS	0.015	**0.017**	0.015	0.019	0.005	0.003	0.006	0.007	0.005	0.009
	Waioeka at Mouth of Gorge	CW	H	HS	0.018	**0.020**	0.018	0.022	0.005	0.003	0.006	0.007	0.005	0.009
	Whakatane at Pekatahi Bridge	CW	H	VA	0.026	**0.032**	0.029	0.036	0.008	0.006	0.011	0.014	0.011	0.018
	Tauranga at Taneatua Bridge	CW	H	VA	0.024	**0.030**	0.027	0.034	0.008	0.006	0.011	0.014	0.011	0.018
	Whirinaki at Galatea	CW	H	VA	0.019	**0.025**	0.022	0.029	0.008	0.006	0.011	0.014	0.011	0.018
	Tarawera at Lake outlet	CW	Lk	VA	0.002									
	Motu at Houpoto	CX	H	HS	0.010	**0.012**	0.010	0.014	0.004	0.002	0.005	0.006	0.004	0.008
	Otara at Browns Bridge	CX	L	HS	0.030	**0.032**	0.027	0.038	0.008	0.003	0.013	0.010	0.005	0.016
	Omanawa at SH29	WW	H	VA	0.022									
	Wairoa at SH2	WW	H	VA	0.010									
	Wairoa d/s Ruahihi Power Station	WW	H	VA	0.012									
	Waiotahe at Toone Rd	WW	L	HS	0.024	**0.027**	0.022	0.034	0.010	0.006	0.016	0.013	0.008	0.020
	Kereu at SH35	WW	L	HS	0.020	**0.023**	0.018	0.030	0.010	0.006	0.016	0.013	0.008	0.020
	Waitekohe at SH2	WW	L	VA	0.003	**0.009**	0.004	0.014	0.009	0.006	0.013	0.015	0.010	0.020
	Raukokore at SH35	WX	H	HS	0.012									
	Ngamuwahine at Old Bridge	WX	H	VA	0.004									
Exotic	Rangitaiki at Te Teko	CW	H	VA	0.017	**0.023**	0.020	0.027	0.008	0.006	0.011	0.014	0.011	0.018
	Rangitaiki at Murupara	CW	H	VA	0.019	**0.025**	0.022	0.029	0.008	0.006	0.011	0.014	0.011	0.018
	Rangitaiki at Matahina Dam	CW	H	VA	0.015	**0.021**	0.018	0.025	0.008	0.006	0.011	0.014	0.011	0.018
	Rangitaiki at Inlet to Aniwhenua Canal	CW	H	VA	0.025	**0.031**	0.028	0.035	0.008	0.006	0.011	0.014	0.011	0.018

	Tarawera at SH30	CW	Lk	VA	0.075									
Exotic	Tarawera at Kawerau Bridge	CW	Lk	VA	0.049									
	Tarawera at Awakaponga	WW	Lk	VA	0.068									
Pasture	Ngongotaha at SH36	CW	H	VA	0.025	**0.031**	0.028	0.035	0.008	0.006	0.011	0.014	0.011	0.018
	Rangitaiki at SH5	CW	H	VA	0.011	**0.017**	0.014	0.021	0.008	0.006	0.011	0.014	0.011	0.018
	Otamatea at Wairere Rd	CW	H	VA	0.021	**0.027**	0.024	0.031	0.008	0.006	0.011	0.014	0.011	0.018
	Puarenga at FRI	CW	H	VA	0.033	**0.039**	0.036	0.043	0.008	0.006	0.011	0.014	0.011	0.018
	Whakatane at Ruatoki	CW	H	VA	0.029	**0.035**	0.032	0.039	0.008	0.006	0.011	0.014	0.011	0.018
	Ohau Channel at SH33	CW	Lk	VA	0.003									
	Kaituna at Maungarangi Rd	CW	Lk	VA	0.018									
	Kaituna at Rotoiti Outlet	CW	Lk	VA	0.004									
	Rocky at Mangatawa Lane	WW	L	Al	0.009	**0.011**	0.007	0.016	0.006	0.002	0.010	0.008	0.004	0.013
	Nukuhou at Glenholme Rd	WW	L	VA	0.019	**0.025**	0.020	0.030	0.009	0.006	0.013	0.015	0.010	0.020
	Pongakawa at SH2	WW	L	VA	0.103	**0.109**	0.104	0.114	0.009	0.006	0.013	0.015	0.010	0.020
	Kopurererua at SH2	WW	L	VA	0.012	**0.018**	0.013	0.023	0.009	0.006	0.013	0.015	0.010	0.020
	Te Mania at SH2	WW	L	VA	0.010	**0.016**	0.011	0.021	0.009	0.006	0.013	0.015	0.010	0.020
	Kopurererua at SH29	WW	L	VA	0.015	**0.021**	0.016	0.026	0.009	0.006	0.013	0.015	0.010	0.020
	Aongatete at SH2	WW	L	VA	0.002	**0.008**	0.003	0.013	0.009	0.006	0.013	0.015	0.010	0.020
	Waiau at Waiau Rd Ford	WW	L	VA	0.010	**0.016**	0.011	0.021	0.009	0.006	0.013	0.015	0.010	0.020
	Pongakawa at Pumphouse	WW	L	VA	0.103	**0.109**	0.104	0.114	0.009	0.006	0.013	0.015	0.010	0.020
	Waimapu 100m d/s SH29	WW	L	VA	0.010	**0.016**	0.011	0.021	0.009	0.006	0.013	0.015	0.010	0.020
	Waitahanui at Otamarakau Marae	WW	L	VA	0.087	**0.093**	0.088	0.098	0.009	0.006	0.013	0.015	0.010	0.020
	Waipapa at Old Highway	WW	L	VA	0.014	**0.020**	0.015	0.025	0.009	0.006	0.013	0.015	0.010	0.020
	Waimapu at Pukemapu Rd	WW	L	VA	0.009	**0.015**	0.010	0.020	0.009	0.006	0.013	0.015	0.010	0.020
	Waitao at Waitao Rd	WW	L	VA	0.006	**0.012**	0.007	0.017	0.009	0.006	0.013	0.015	0.010	0.020
	Pongakawa at Old Coach Rd	WW	L	VA	0.105	**0.111**	0.106	0.116	0.009	0.006	0.013	0.015	0.010	0.020
	Kaituna at Te Matai	WW	Lk	VA	0.027									
	Kaituna at Te Tumu	WW	Lk	VA	0.022									
	Te Rereatukahia at SH2	WX	L	VA	0.004									
	Tuapiro at Hikurangi Rd	WX	L	VA	0.004									
	Uretara at Henry Rd Ford	WX	L	VA	0.003									

Native, Native vegetation; Exotic, Exotic forest