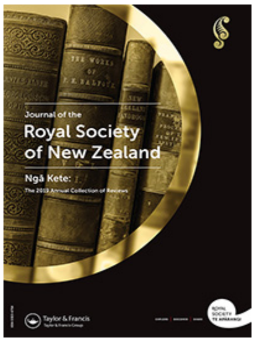

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REVIEW ARTICLE

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Why are median phosphorus concentrations improving in New Zealand streams and rivers?

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ABSTRACT

The enrichment of phosphorus (P) in streams and rivers can impair water quality, but concentrations have decreased. We found little evidence that this improvement was caused by a decrease in soil Olsen P concentrations or imported P (e.g. fertiliser), a change to low water-soluble P fertilisers, or that greater nitrate loads were assimilating P from groundwater or sediments. Possible causes of improvement were that land use change had decreased erosion, more nitrogen fertiliser use was assimilating soil P, and a greater awareness of P as an environmental issue. However, the most probable causes were that strategies were mitigating P loss from land, guidelines were directing where to best use strategies, and policy instruments were including P management. These findings support the development and implementation of mitigations, supported by voluntary guidelines and regulation. However, our findings can be strengthened if data are referenced to equivalent, and finer spatial and temporal scales.

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Critical source areas; fertiliser; good management practices; guidelines; policy

Introduction

Water quality in New Zealand receives considerable scholarly and public attention (Hughey et al. 2016). Much of this attention has drawn a link between declining water quality and the proportion of high-intensity land uses in a catchment such as intensively grazed pasture (Wilcock 1986; Wilcock, Monaghan, Quinn, et al. 2013; Snelder et al. 2017). The enrichment of surface freshwaters with P can stimulate algal growth (some species are toxic) leading to the loss of oxygen when algae die, fish kills, and impairment of water for recreation, drinking, and industrial and agricultural uses (Carpenter et al. 1998). Previous analyses of water quality data in New Zealand has shown the majority of sites examined by regional authorities and science providers have P concentrations that are likely to limit periphyton growth (McDowell et al. 2009). Recent national policy has set an objective that limits chlorophyll-a concentrations to <200 mg m^{-2}

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(Ministry for the Environment 2017). Chlorophyll-a is a measure of periphyton growth, which is in turn driven by the concentration of the limiting (or co-limiting) nutrient, nitrogen (N) or P.

Trends in the concentrations of various water quality indicators including filterable reactive phosphorus (FRP, also called dissolved reactive P) and total P (TP) are developed as part of regional and national reporting requirements (Ministry for the Environment and Statistics New Zealand 2017a). The analysis of data from 1994 to 2013 showed that for 145 monitoring sites in catchments dominated by intensively grazed pasture, median FRP concentrations were decreasing at 46% of locations, increasing at 21% of locations and had showed no change elsewhere (Ministry for the Environment and Statistics New Zealand 2017a, 2017b). However, when examined at 277 sites between 2004 and 2013 (more had subsequently been established), median FRP concentrations were decreasing at 57% of locations, and increasing at 15% of locations; 29% remained unchanged. For the 159 sites dominated by intensively grazed pasture monitored for TP between 1994 and 2013, 41% of sites were improving (21% were worsening), while for the 304 sites monitored between 2004 and 2013, 65% were improving. This increase in sites exhibiting an improvement in water quality occurred despite an increase in national dairy cow numbers by 26%, and the continued expansion of dairying into new areas commonly used for sheep farming; sheep numbers decreased by 22% over the same period (Statistics New Zealand 2018).

Reasons for increases or decreases in P concentrations can be implied, but the strength of any reasoning is dependent on factors such as flow-paths, land use practices and the quality and amount of information over spatial and temporal scales (Álvarez et al. 2017). For example, at a small or sub-catchment scale, streamflow may be dominated by surface flows that move quickly from the land into the stream (Woodward et al. 2016). However, at a large catchment or regional scale, more streamflow is sourced from deeper and potentially older water (Morgenstern and Daughney 2012). This means that while regular sampling for P at a small scale may help pin-point the effect of certain land use practices on streamflow P concentrations (Monaghan et al. 2009). At a larger scale the ability to ascribe P concentration trends to land use practices is compromised by our ability to monitor those practices at enough sites and water that is sourced from a range of flow-paths and ages.

Despite these challenges, an ever-increasing amount of data is available. We are now at a stage where we can provide some guidance on the strength of factors (e.g. land use practices) put forward in the scientific and public literature as likely causes of P trends in streamflow (Larned et al. 2016; McDowell et al. 2016; Ministry for the Environment and Statistics New Zealand 2017a). Such factors include, but are not limited to: (1) land use change; (2) a decrease in imported P as fertilisers and feed; (3) a change in P fertiliser form; (4) a decrease in soil Olsen P concentrations; (5) greater assimilation of P in soils; (6) greater assimilation of P in groundwaters and in-stream; (7) an increase in the use of P mitigations; (8) better awareness and education of rural professionals of P; and (9) the use of policy instruments. This paper is a review of the evidence for each factor. In this examination we note a couple of caveats. Firstly, owing to the unique nature of our farming systems and catchments, the majority of this evidence comes from New Zealand studies, but is augmented where applicable by international findings. Secondly, while the trends in P concentrations are spatially referenced to a catchment, unlike some other studies (Santos et al. 2017), the majority of available data is not. We were therefore unable to use complex statistical analyses to show why trends were occurring, but have provided a semi-quantitative estimate of the strength of each factor classified as: unlikely, possible but unclear, or probably causing a decrease FRP and TP concentrations across New Zealand streams and rivers.

Materials and methods

In our national study a large number of spatial and temporal databases were used. The coverage and limitations of these databases are outlined below. Given the format of the paper as a review, supported by semi-quantitative data, and the limitations of many of the databases, the majority of methods used either descriptive statistics or used spatially- or temporally-referenced correlations of log-transformed data (where appropriate) at the finest possible scale. Owing to the simplicity of these methods, we have not provided a lengthy, and likely repetitive, statement of how they were used to test each of the factors (1–9). Instead, we provide a brief summary here and when discussing each factor.

The datasets and associated methods used to examine the likely strength of each factor (1–9) are listed below.

(1) Land use change: data from a variety of datasets (Table 1) was used to derive land cover, land use and stocking rates within each of the catchments. Flow-weighted trends in FRP and TP concentrations were extracted for the periods between 1994 and 2013 and 2004 to 2013 (Larned, Scarsbrook, et al. 2004; Ministry for the Environment and Statistics New Zealand 2017a). Trends were also extracted for clarity as a likely indicator of sediment and potentially particulate-P loss (Davies-Colley and Smith 2001; Ekholm and Krogerus 2003). Data for sediment or turbidity were not available. Coverage for land use and stocking rates was better for 2004–2013 than 1994–2003. To improve the reporting of land use data and avoid large areas where there were no data, the Land Cover Database was intersected with AgriBase, a national spatial database of more detailed land use (e.g. farm type, size, animal numbers, plated areas for orchards, crops, forestry, etc.). AgriBase data were used in the first instance, but reported as ‘high-producing exotic grassland’, as

Table 1. Summary of the types, sources, temporal and spatial scales of biophysical data used in the study.

Data type	Data source	Years covered	Spatial scale	References
Land cover	Land Cover Database	1996/97, 2001/2002, 2008/2009, 2012/2013	Sub-catchment	Landcare Research (2017)
Pastoral land use	AgriBase – AsureQuality	1999, 2002, 2006, 2013	Farm[a]	Sanson (2005)
Numbers of different farm types	Statistics New Zealand	1996, 1999, 2003, 2007, 2012	Regional	Statistics New Zealand (2018)
Fertiliser N and P sales	Fertiliser Association of New Zealand, Ballance Agri-Nutrients, Ravensdown, Statistics New Zealand	1994–2015 (for P) 2002, 2012 (for N)	Regional	Statistics New Zealand (2018)
Imported P in dairy cow feed	DairyNZ	1994–2015	National	DairyNZ Economics Group (2016)
Soil Olsen P concentrations	Analytical Research Laboratories, Eurofins, Hill Laboratories, Soil Fertility Service	2001–2014 2000–2014 2008–2014 1994–2002	Regional	

[a] Incomplete as contributions are voluntary.

per the LCDB, where no data existed. A similar approach was used by Daigneault and Elliott (2017) in correlating national scale water quality contaminant loads with land use. We had access to data from AgriBase for 1999, 2002, 2006 and 2013. Data were not available to exactly match the 1994–2003 and 2004–2013 years.

Data for the change in catchment areas (in ha and percentage changes) between 1999–2013 and 2006 and 2013 in different land uses (dairy, deer, forestry, native bush, and sheep and beef) were compared to the percentage magnitude change in significant FRP, TP and clarity trends in corresponding catchments using ArcGIS v10.6 and Pearson correlation coefficients via Genstat v17 (Genstat Committee 2015). Percentages were used to normalise data in place of absolute magnitude differences which were skewed. Significant trends for flow-weighted FRP, and TP concentrations were calculated as a significant increase or decrease in median FRP or TP concentration divided by the commensurate median FRP or TP concentration for a site. Analysis involved deriving correlations between trends for P concentrations and trends for land use changes (e.g. stock numbers and coverage) at a catchment scale.

(2) A decrease in imported P as fertilisers and feed: data were sourced for P fertiliser sales from Ballance Agri-Nutrients and Ravensdown fertiliser cooperatives at a regional level, and compared via Pearson correlation analysis to area-weighted trends in P concentrations. These companies together account for about 98% of fertiliser sales in New Zealand (Fertiliser Association of New Zealand 2018). Additional data for the quantity of P imported in animal feed was obtained from the DairyNZ Economics Group (2016).

(3) A change in P fertiliser form: data were sourced from Ballance Agri-Nutrients and Ravensdown fertiliser cooperatives at a regional level for the sales of superphosphate and reactive phosphate rock (RPR) and compared via Pearson correlation analysis to area-weighted trends in P concentrations.

(4) A decrease in soil Olsen P concentrations: data were sourced from all the major commercial soil testing labs in New Zealand (Table 1). Data were linked to a region, and sometimes to the sub-region or catchment scale. However, for consistency, we assigned data from all sources to regions. Data were presented as a significant change in the slope of a regression between log-transformed Olsen P concentrations and time (year) by region and also as the percentage of sites exceeding an agronomic optimum for pasture production (Morton and Roberts 1999; Roberts and Morton 2009) in Sedimentary, Volcanic, Pumice and Peat soils (20, 35, 45 and 45 mg P L^{-1}, respectively). Pastoral land is by far the dominant land use in New Zealand, covering approximately three-quarters of farm land (Statistics New Zealand 2018).

(5) Greater assimilation of P in soils: data for N fertiliser applied at a regional scale were sourced from the Agricultural Census for 2002 and 2012 (Statistics New Zealand 2018) and compared via Pearson correlation analysis to area-weighted trends in P concentrations.

(6) Greater assimilation of P in groundwaters and in-stream: data were sourced from the National Groundwater Monitoring Programme for 540 sites around the country. This dataset was initially compiled by Daughney and Reeves (2006) and Daughney et al. (2012), but subsequently updated to 2015 by the Ministry for the Environment and GNS Science. The analysis involved determining if there was a significant correlation (Pearson) between a trend in FRP concentrations in surface waters and the nearest groundwater sites within a 5 km radius.

(7) An increase in the use of P mitigations: data were sourced from the 2013–2017 surveys of Rural Decision Makers who canvassed 1500 (2013)–4500 (2017) farmers nationally (Manaaki Whenua: Landcare Research 2018), and the Clean Streams Dairy Accord and the Sustainable Dairying Water Accord to gauge the use of mitigations on dairy farms (Sanson and Baxter 2011; Dairy Environment Leadership Group 2013; Ministry for Primary Industries 2013; DairyNZ 2018). These data were compared via Pearson correlation analysis to area-weighted trends in P concentrations

(8) Better awareness and education of rural professionals: demographic data were sourced from Massey University's professional development courses in nutrient management. These courses are required of all staff at Ballance Agri-Nutrients and Ravensdown fertiliser cooperatives, and are well attended by Regional Council staff and rural advisors. We compared the number of rural professionals to frequency of improving trends by region.

(9) An increased use of policy instruments: data were sourced from internet search engines, correspondence with the primary sector, and government agencies on the timing and geographic coverage of national and regional scale policy documents. A narrative is given of their likely impact assuming effective compliance by land owners and users to policy.

Results and discussion

Processes involved in P loss from land to water

The loss of P from land to streams and rivers is a function of the availability of P to loss, a transport pathway to get P to streams and rivers, and intervening attenuation processes along the transport pathway that may decrease P losses. All three processes can be influenced by landuse and land management practices. However, of the land available for primary production in New Zealand (52% of total area), three-quarters is used for grazed pastoral agriculture (Ministry for the Environment and Statistics New Zealand 2018). Hence, the following outline of processes focuses on pastoral agriculture, but soil- and fertiliser-P losses are also relevant to other land uses.

Sources of P loss from land to water include: the soil, fertiliser, plant residues and animal dung deposited directly or as farm dairy shed effluent (FDE) (Mundy et al. 2003; Curran-Cournane et al. 2011). As P surpluses increase, it is likely that soil P concentrations increase. Soil P-enrichment increases the availability of P to loss, especially in soils with low anion storage capacity (ASC) that have few Al- and Fe-oxides to sorb and retain P (McDowell and Condron 2004). The erosion of particulate-associated P can be enhanced by soil compaction and pugging following treading by grazing animals (Bilotta et al. 2007). Direct loss of P can also occur via animal dung (not urine) and fertiliser, although the availability of P in fertiliser is proportional to the fertiliser's water solubility (Hart et al. 2004). Plant residues can be a source of P for a short time after forage is grazed by ruminants. The availability of P for loss by dung deposition decline exponentially with time as a crust forms on the dung, thus impairing interaction with rainfall (Smith et al. 2001).

Transport pathways include runoff, encapsulating surface runoff and subsurface flow (viz. interflow), and drainage to groundwater. For surface runoff, P losses can occur via infiltration-excess and saturation-excess mechanisms. Under infiltration-excess conditions, the infiltration capacity of the soil is exceeded resulting in surface runoff. In New Zealand, this usually occurs under high-intensity rainfall or hydrophobic soil conditions at any time of year (Bretherton et al. 2011; Müller et al. 2018), whereas saturation-excess surface runoff only occurs when soils are saturated (largely in winter and spring) resulting in any excess rainfall running-off. Due to the energy of high-intensity rainfall events, infiltration-excess surface runoff can contain more particulate-P than saturation-excess surface runoff (Buda et al. 2009). Due to topography, areas affected by saturation-excess surface runoff are generally located near the stream channel and expand and contract in response to rainfall events and evapotranspiration. Timing wise, most surface runoff and P loss occurs in large events in winter and spring due to saturation-excess surface runoff (Sharpley et al. 2008). However, losses at this time are less likely to cause algal growth (unless not flushed from the system) compared to losses in summer when low flows and higher temperatures facilitate growth.

Attenuation (i.e. removal) of P in surface runoff can occur along the flow path as heavy particulates settle-out, but can also occur when filtered out by plants, for example, at the point where runoff from an arable field intersects with a buffer strip. Note however that such filtration will not occur if the runoff originates in pasture where few coarse particles would be produced and likely filtered out by the sward before flow reaches the buffer strip (Thomas et al. 2018). The leaching of P through interflow and to groundwater is generally greatly attenuated by sorption and filtration in the vadose zone. An exception is P that is transported by artificial drainage. Here, macropores can provide a rapid conduit between surface P and streams via subsurface drains, although some filtration can occur resulting in a greater proportion of P being lost dissolved over particulate forms than likely in surface runoff (Monaghan et al. 2016). Another exception occurs where moderate amounts of P are applied to low P sorption capacity soils (measured as ASC) and leached by regular rainfall or irrigation. If draining into an aquifer of low ASC (e.g. sand and gravel), groundwater can become P-enriched and enriched baseflow in nearby streams (McDowell et al. 2015).

(1) *Land use change*

A wealth of literature indicates that the loss of P to water is correlated to the presence and magnitude of intensive land uses (Julian et al. 2017; Meyfroidt 2017). However, our data yielded few significant correlations between the percentage changes in either stock numbers or the area within a catchment of a stock class and the percentage change of FRP or TP trends (Table 2). This result doesn't exclude the possibility that some sites may show a strong association with a change in stock numbers or the catchment percentage area occupied by a stock class, but does suggest that other factors such as climate, slope, land management (see the section 'Process involved in P loss to water') likely play a stronger role in influencing P concentrations.

Some, weak, correlations were found between a decrease in sheep numbers and TP (1999–2013) or the area occupied by sheep and the percentage change in FRP concentrations. Interestingly, an increase in the area occupied by dairy was also weakly correlated to an increase in TP between 2006 and 2013 (Table 2). Due to their weak association it is difficult to draw substantive conclusions on the effect of land use change on P losses. However, as the P loss from sheep-farmed land is usually less than from dairy-farmed land (McDowell and Wilcock 2008), it is sensible that an increase in sheep numbers, at the expense of other stock classes, could result in a decrease in either FRP or TP if

Table 2. Mean percentage change (and standard deviation) in catchment stock numbers and area for different stock classes for the periods 1999–2013 and 2006–2013 and their respective comparison (Pearson correlation coefficient) to trends between 1994–2013 and 2004–2013 for either in the percentage change in median annual FRP or TP concentrations and clarity.

Attribute	Land use	Mean % change	Standard deviation of change	Correlation to FRP trend	Correlation to TP trend	Correlation to clarity trend
Change in stock numbers 1999–2013 (%)	Beef	47.4	9.1	−0.07	−0.10	0.15
	Dairy	209.9	44.6	0.09	0.01	0.01
	Deer	152.7	35.1	0.08	0.08	−0.01
	Sheep	152.2	49.8	−0.10	−0.12*	−0.02
Change in catchment area 1999–2013 (%)	Beef	−2.4	0.3	−0.04	−0.03	0.06
	Dairy	1.8	0.3	0.12	0.07	−0.02
	Deer	−0.2	0.1	0.03	−0.06	0.05
	Sheep	−3.4	0.4	−0.14*	0.05	−0.18*
	Sheep/Beef	9.3	0.5	0.09	0.02	0.06
Change in stock numbers 2006–2013 (%)	Beef	−3.4	1.0	−0.01	−0.10	0.06
	Dairy	110.5	55.1	−0.02	−0.04	0.06
	Deer	7.6	12.0	0.02	0.00	−0.08
	Sheep	92.8	51.9	0.06	0.09	−0.08
Change in catchment area 2006–2013 (%)	Beef	0.0	0.1	0.04	0.03	0.05
	Dairy	0.8	0.2	0.07	0.12*	−0.06
	Deer	−0.1	0.0	−0.07	−0.03	−0.02
	Sheep	−0.4	0.1	−0.10	−0.03	−0.18*
	Sheep/Beef	0.0	0.1	−0.06	−0.09	−0.04

* Indicates a weak ($P < .05$), but significant correlation.

there is little carry-over (or legacy) or past land uses. Similarly, an increase in the area under dairying is expected to result in an increase in TP losses, especially since the number of dairy cattle increased more in the 2006–2013 period (3.9–4.9M = 143,000 cows p.a.) than over the 1999–2013 period (3.2–4.9M = 121,000 cows p.a.) (DairyNZ 2014). Indeed, using the National River Water Quality network of 77 large scale catchments Julian et al. (2017) was able to show weak associations with changing nutrient concentrations and stock numbers, but found other factors such as soil type were equally important in many of the catchments.

Due to the variability of soil type and factors important in erosion processes (e.g. climate and slope), clarity cannot be used to accurately predict suspended sediment loads (Davies-Colley and Smith 2001), but can be helpful in isolating trends at a site where landforms and erosion processes don't change, but management does. We isolated sites that showed decreases in FRP or TP concentrations and increases in clarity (Table 3). More sites, and a greater percentage of these sites, exhibited this phenomenon in 2004–2013 than from 1994 to 2013 (Table 3). The phenomenon was more frequent for FRP than TP (Table 3). This could reflect the fact that clarity is sensitive to the presence of fine particles, which have a greater affinity for sorbing P than coarse particles (Brennan et al. 2017). However, the geographic distribution of sites exhibiting this phenomenon also changed between 1994–2013 and 2003–2013. In general, the areas exhibiting the phenomenon over 1994–2013 were more likely to be associated with upland drystock or forestry than in 2004–2013 (Figure 1).

The incidence and increasing percentage of sites exhibiting improving trends in clarity and P concentrations would suggest that efforts to promote practices that prevent erosion may have also decreased P losses. Weak associations between changing P concentrations

Table 3. Count of sites for the 1994–2013 and 2004–2013 periods showing an increasing or decreasing trend in FRP or TP concentrations, and the count of sites for the same periods exhibiting a decreasing trend in FRP or TP concentrations and increasing clarity at the same site. Sites showing indeterminate trends were not considered.

Analyte and period	Increasing	Decreasing	Decreasing P and increasing clarity
FRP			
1994–2013	142	71	27 (38%)[a]
–2013	189	122	75 (61%)
TP			
–2013	46	99	26 (26%)
2004–2013	25	251	104 (41%)

[a] Corresponds to the percentage of sites with decreasing FRP or TP and increasing clarity for the period.

and changes in catchment land use suggest that a greater influence of P concentrations has been land practices, not the magnitude of land use change. Using 'trees on farms' (e.g. spaced – poplar planting) has been highly successful in preventing soil erosion on hill country sheep and beef farms (McIvor et al. 2011) without changing land use designation and will be discussed in Section 7. It is therefore possible, but unclear if changes in land use caused decreases in P concentrations.

(2) *A decrease in imported P as fertilisers and feed*

The potential for P loss parallels the magnitude of soil P enrichment (Heckrath et al. 1995). The enrichment and maintenance of soil P concentrations is achieved through the application of P (Roberts and Morton 2009). In grazed pastoral systems, most of this P comes

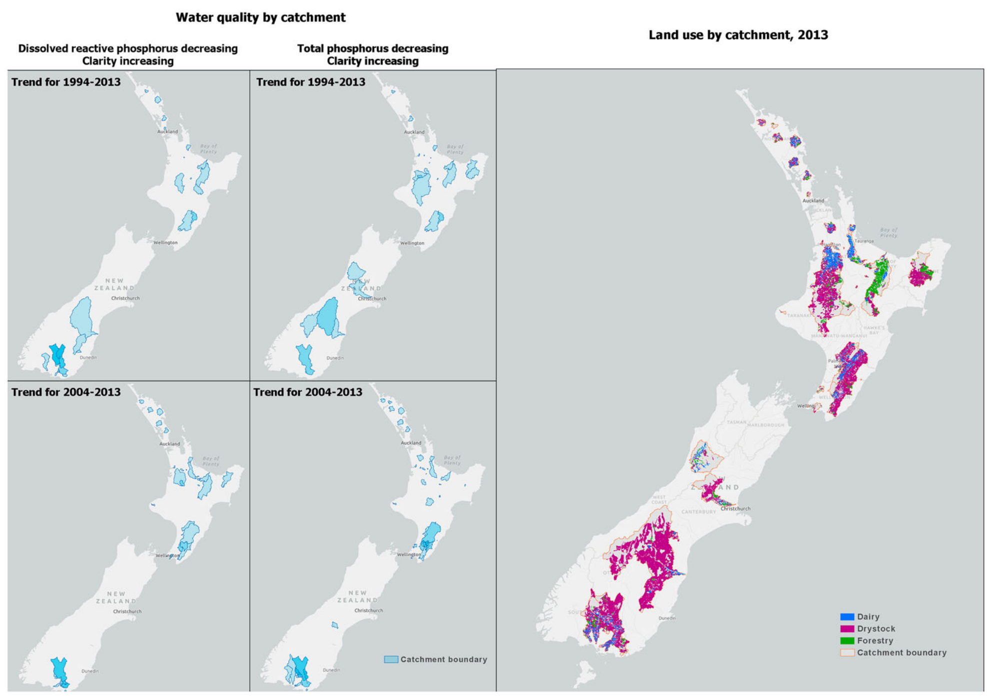

Figure 1. Co-location of trends for decreasing FRP or TP and increasing clarity for 1994–2013 and 2004–2013 and land use in each catchment for 2013. Blank areas within a catchment are most likely to be native forest or mountains.

from fertiliser with a small amount coming via the dung of animals eating imported feed; feed-P has increased since 2007 (Figure 2) due to the use of palm kernel extract on dairy farms (Robertson 2010; DairyNZ Economics Group 2016). The mean annual sales volumes of P (as superphosphate plus RPR) and of imported feed for the period 2004–2013 was on average 15% less than in the period 1995–2004. However, this difference was not significant (t-test, $P > .05$). It is therefore unlikely that a decrease in imported P contributed to decreases in in-stream P concentrations (Figure 2). However, the power of our analysis could be improved if imported P data were available at a finer (e.g. catchment) scale.

(3) *A change in P fertiliser form*

The contribution of fertiliser to total farm P losses from grazed pastoral agriculture is estimated to be around 10% if good practices are followed such as applying in summer when runoff is less likely (McDowell, Nash, et al. 2007). However, there is increasing evidence that this proportion is greater under normal practice, and can be up to 90% if poor practice is followed (Hart et al. 2004). Poor practice includes the application of soluble P just prior to flood irrigation or a rainfall event where runoff is expected. Under normal practice where P is applied in summer, fertiliser-P can be lost via short, intense storms that produce runoff via infiltration-excess processes which may be exacerbated by hydrophobicity (Doerr et al. 2003; Müller et al. 2018). In addition, where P is applied in high rainfall environments, where runoff cannot be avoided, fertiliser P losses can be 3–5 kg P ha^{-1} yr^{-1} (McDowell 2010).

The availability of P for loss is dependent on the water solubility of P applied. Soluble P, derived from superphosphate (90–97% water soluble), is available for loss in large amounts for around 7–14 days after application, and significantly greater than background concentrations for 60 days after application (McDowell et al. 2003). Serpentine superphosphate and dicalcic superphosphate contain around 10–15% water-soluble P. However,

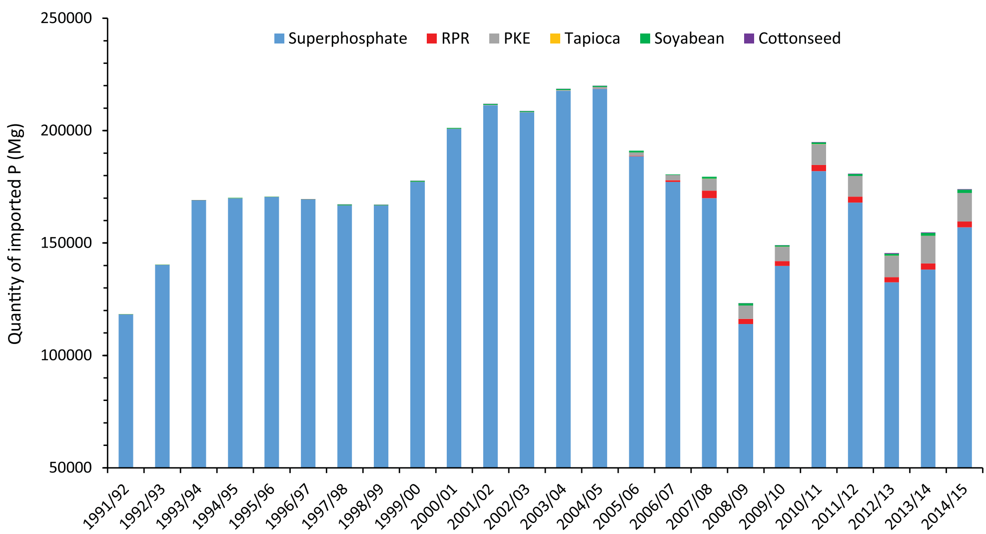

Figure 2. Annual quantity (Mg) of imported P given as sales of superphosphate and RPR and imported feed (PKE = Palm Kernel Extract) using their average per cent P concentrations (DairyNZ 2018).

RPR only contains around 1–2% water-soluble P with its remaining P slowly solubilising into the soil solution over several years. This slow-release characteristic greatly reduces the availability of P from RPR, and to a lesser extend serpentine superphosphate and dicalcic superphosphate, to runoff events soon after application (McDowell and Catto 2005). Hart et al. (2004) reviewed the literature and hypothesised that P losses would be lower from plots and farms with RPR applied than those with superphosphate applied due to a lower water solubility. Research at a catchment scale has subsequently showed that in two catchments receiving the same rate of P the RPR treated catchment lost on average 38% less FRP than the superphosphate treated catchment (McDowell et al. 2010). Although, applied in June, when the loss of P from P-applied would likely be greatest, this data suggested that the application of a lower water-soluble P product is effective at decreasing P losses.

Sales of RPR sit at between 1% and 2% of P sold in New Zealand. Similar levels of serpentine superphosphate are sold, although the data are unclear, while sales of dicalcic superphosphate are consistently <1% annually. Low sales may be due to poor recognition of its environmental benefit, associated with reduced P losses compared to superphosphate, but is more likely due to the unreliable availability of good quality RPR and climate and soil conditions that constrain its use. For RPR to maintain pasture production at the same rate as superphosphate rainfall should be >800 mm per year and soil pH < 6.0 (Sinclair et al. 1993). Furthermore, due to poor solubility transitioning from superphosphate requires RPR to be gradually introduced, replacing 33% of superphosphate in the first year, 66% of superphosphate in year 2 and all superphosphate from thereon in (Sinclair et al. 1990). Slow reactivity also means that RPR is also not suitable for capital applications of P (Sinclair et al. 1990). Low sales from 2004 to 2013 suggest a change to RPR is unlikely to be a reason for decreased P losses.

(4) *Soil Olsen P concentrations have decreased*

Soil Olsen P data from 2002 to 2014 (N ~ 500,000 samples) were collected from Analytical Research Laboratories, Eurofins, Hill Laboratories, and the Soil Fertility Service (which ceased business in the mid-1990s). The potential for P loss increases with soil Olsen P concentrations, and where soil P sorption capacity is limited: capacity is inversely proportional to ASC (McDowell and Condron 2004). In the absence of ASC data, the 'rule of thumb' is that the potential for P losses increases markedly when Olsen P concentrations exceed their agronomic optimum (Morton et al. 2003). Using data from Eurofins, Hill Laboratories and the Soil Fertility Service, who recorded samples by their soil grouping as Volcanic, Pumice, Sedimentary and Peat, Wheeler et al. (2004) showed that the proportion of sites in excess of the agronomic optimum (Olsen P) increased from 1988–1996 to 1997–2001. Data from 2002 to 2014 showed a decrease in the national proportion of samples in excess of the agronomic optimum for Volcanic and Pumice soils, but enrichment for Sedimentary and Peat soils (Figure 3). However, additional regression analysis (also including the ARL data) showed a small increase in annual mean Olsen P concentrations for most regions from 2002 to 2014 (Table 4). This suggests that although there were fewer samples in excess of an agronomic optimum, there was a general increase in Olsen P concentrations both above and below the agronomic optimum. Olsen P concentrations have not decreased and is therefore unlikely to be a cause of declining P concentrations in-stream.

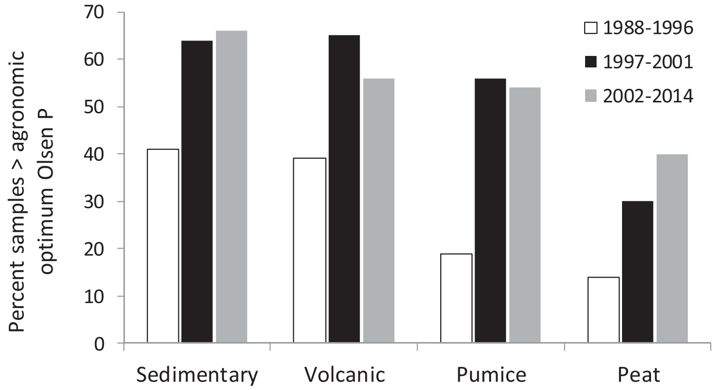

Figure 3. The relative percentage of samples in excess of the agronomic optimum in Olsen P concentration for pasture production (20, 35, 45 and 45 mg P L^{-1} for Sedimentary, Volcanic, Pumice and Peat soils, respectively) submitted to Eurofins and Hills laboratories (n ∼ 500,000) between 1988–1996, 1997–2001 and 2002–2014. Data for 1988–1996 and 1997–2001 is from Wheeler et al. (2004).

(5) *The greater assimilation of P in soils and aquifers*

The addition of N fertiliser to soils boosts pasture yield, requiring more P which the plant obtains from the soil (Haynes and Williams 1993). As the soil solution is the conduit through which P must be sequestered, reduced concentrations of P in the soil solution also results in lower leaching losses. In a two-year trial on three intact pasture soils of varying ASC, Dodd et al. (2014) found that the application of 150–300 kg N ha^{-1} yr^{-1} resulted in a decrease in annual FRP load in leachate by 53–76% in two soils of moderate ASC (51–53%) compared to a control receiving no N (Figure 4). No decrease was found in the higher ASC soil (72%), whose greater sorption led to more P in the solid phase and less leached.

Agricultural censuses data for the regional use of N fertiliser in 2002 and 2012 are given in Table 5. Although the proportional mass of N fertiliser applied has increased, especially in some regions, the growth in use doesn't correlate well with the frequency of decreasing trends in P concentrations (Table 3). For example, some regions like Canterbury exhibited

Table 4. Trends in topsoil Olsen P concentrations (mg L^{-1}) between 2002 and 2014 and submitted to ARL, Eurofins and Hill Laboratories.

Region	Number of samples	Trend (mg L^{-1} Olsen P yr^{-1})	Significant
Auckland	8627	0.28	Y
Bay of Plenty	14,030	0.19	Y
Canterbury	111,348	0.08	Y
Gisborne	3434	1.15	Y
Hawkes Bay	19,538	0.45	Y
Manawatu/Wanganui	37,982	0.80	Y
Marlborough	5064	0.07	N
Northland	16,462	0.50	Y
Otago	58,679	0.02	N
Southland	96,596	0.22	Y
Tasman	8875	0.85	Y
Taranaki	16,391	0.05	N
Waikato	63,339	0.59	Y
Wellington	7106	0.29	Y
West Coast	11,652	0.39	Y

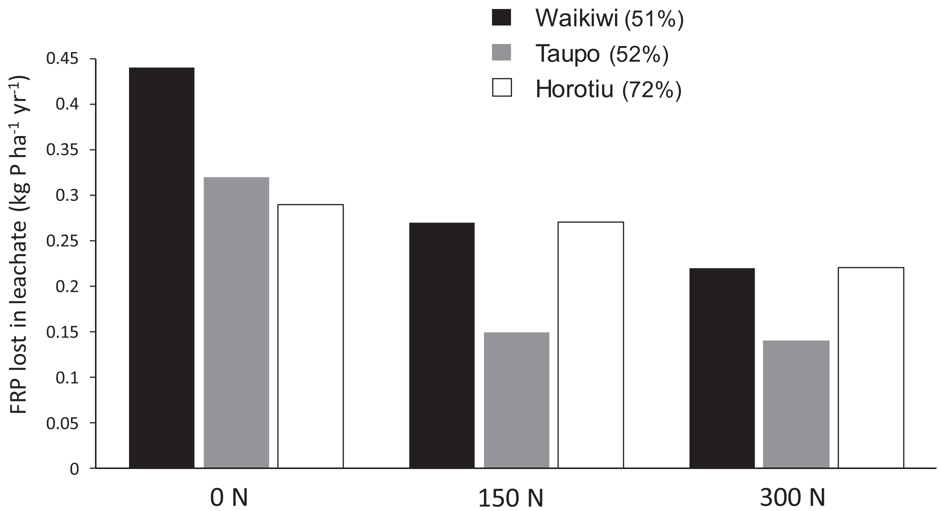

Figure 4. Mean annual load of FRP lost in leachate from the top 30-cm of three pasture soils of varying ASC (51–72%) receiving either no N or 150 or 300 kg N ha^{-1} yr^{-1} as urea (Dodd et al. 2014).

a strong increase in N applied (93%) and had a large number of sites showing decreases in FRP (31) or TP (22), but Auckland also had many sites showing decreases in FRP (17) or TP (19), but N applied decreased. It is therefore possible but unclear if an increase in N sales contributed to decreases in P concentrations.

(6) *The greater assimilation of P in aquifers and in-stream*

Nitrate (NO_3) is widely recognised as a powerful oxidant (Galloway et al. 2003). Increasing loads of NO_3 can therefore drive a shift in the redox status of soils, aquifer and river sediments. Greater and more sustained NO_3 loading could lower FRP concentrations in groundwater and in-streams via two key mechanisms, buffering and oxidation. Some evidence shows that greater NO_3 loads can buffer (viz. limit) the microbially-mediated

Table 5. Data for the regional tonnage of N fertiliser applied in New Zealand in 2002 and the percentage change in tonnes applied by 2012.

Region	Mass of N sold (Mg) in 2002	Change 2012 relative to 2002 (%)	Number of sites with decreasing FRP (TP) concentrations
Auckland	4638	−19	17 (19)
Bay of Plenty	11,358	6	13 (6)
Canterbury	42,228	93	31 (22)
Gisborne	2834	1	1 (2)
Hawkes Bay	8901	−8	12 (25)
Manawatu/ Wanganui	20,757	2	6 (4)
Marlborough	1758	−24	1 (1)
Northland	11,446	9	10 (13)
Otago	12,016	49	5 (14)
Southland	17,155	79	16 (21)
Tasman	3142	3	4 (0)
Taranaki	13,264	49	3 (3)
Waikato	50,970	14	63 (87)
Wellington	5158	3	19 (39)
West Coast	6190	92	0 (0)

reductive dissolution of the oxides/oxy hydroxides of Mn and Fe in anaerobic sediments thereby maintaining their P sorption capacity (Surridge et al. 2007). In addition, as a strong oxidant, greater NO_3 loadings can increase the direct oxidation of ferrous (Fe^{II}) to P-sorbing ferric (Fe^{III}) iron in aquifers (Uhlmann and Paul 1994).

Table 6 shows the relative percentage of increasing and decreasing trends for NO_3 and FRP, respectively for streams and rivers, and in groundwater – taken in this instance as an important contributor of flow and P to streamflow. Increasing NO_3^- concentrations in groundwater or streams and rivers was only mirrored by a decrease in FRP concentrations at 1% and 4% of sites, respectively. It is possible that NO_3 concentrations in groundwater or streams and rivers were insufficient to decrease FRP concentrations due to poor connectivity to NO_3-rich topsoils. However, the scarcity of sites does not support the hypothesis that the enrichment of NO_3 concentrations in groundwaters or streams and rivers increased the removal of FRP via sorption to Fe and Mn oxides.

In addition to influencing chemical processes, NO_3 is also a stimulant of periphyton growth (Dodds and Smith 2016). The frequency of periphyton outbreaks and biomass has been increasing in New Zealand streams and rivers (Snelder et al. 2013), and have paralleled an increase of NO_3 losses to streams and rivers over the last 20 years (Ministry for the Environment and Statistics New Zealand 2017a). This could increase the demand for P by periphyton. An increased demand would be seen as a decrease in FRP, which is more bioavailable than particulate and organic P forms (Ekholm and Krogerus 2003). However, research has shown that during the growing season and under period of stable flow, periphyton growth in New Zealand streams and rivers is more often limited by P than by N (Larned, Nikora, et al. 2004; McDowell et al. 2009). Moreover, the data in Table 6 did not support the hypothesis that increasing trends in NO_3 were responsible for decreased FRP concentrations in streams and rivers. However, this hypothesis should not be discounted as concentrations and trends in nutrient concentrations could also be masked by whether or not periphyton are actively growing and taking up N or P or are sourcing P from sediments (Wood et al. 2015).

(7) *An increase in the number and use of P mitigations*

Table 7 shows when strategies to mitigate P losses from New Zealand enterprises, and their cost-effectiveness, were published. The majority of these strategies were developed and assessed in 2000–2010. General advice on these strategies is that a user should only consider those strategies that are applicable to their enterprise, implement the most cost-effective first, and use them in critical source areas to improve cost-effectiveness (Bailey et al. 2013; Gooday et al. 2014; McDowell 2014; Vinten et al. 2017). Critical source areas are small areas of a farm or catchment that account for disproportionately large amounts of contaminant (e.g. P) losses because they are hydrologically active and are

Table 6. The relative percentage of sites exhibiting increasing trends in nitrate (NO_3), decreasing trends in FRP and their co-location in streams and rivers or groundwater for data for 2004–2013.

Trend	Groundwater	Streams and rivers
Increasing NO_3	82	51
Decreasing FRP	17	45
Collocation of increasing N with decreasing P	1	4

Table 7. Range of strategies used in management, as an amendment or at the edge of the field to decrease P losses to water, their effectiveness, cost and citation noting the year that their cost-effectiveness was assessed.

Strategy		Main targeted P form(s)	Effectiveness (% TP decrease)	Cost – Range (USD $ kg^{-1} P conserved)[a]	References
Optimum soil test P	Management	Filterable and Particulate	5–20	(Highly cost-effective)[b]	Morton et al. (2003)
Low solubility P fertiliser		Filterable and Particulate	0–20	0–30	McDowell et al. (2010)
Low P near stream areas and P efficient pastures		Filterable and Particulate	26–46	(Highly cost-effective)[c]	McDowell et al. (2014)
Stream fencing		Filterable and Particulate	10–30	2–45	McDowell (2008)
Restricted grazing of cropland		Particulate	30–50	30–200	McDowell et al. (2005)
Delayed strategic grazing of cropland gullies		Particulate	65	20 –80	Monaghan et al. (2017)
Greater effluent pond storage / application area		Filterable and Particulate	10–30	3–50	Houlbrooke et al. (2004)
Flood irrigation management[d]		Filterable and Particulate	40–60	3–300	Houlbrooke et al. (2008)
Low rate effluent application to land		Filterable and Particulate	10–30	8–55	Monaghan et al. (2010)
P-sorptive backfill in tile drains	Amendment	Filterable and Particulate	50	30–110	McDowell et al. (2008)
Post wetland drainage filter		Filterable	49–98	80–200[e]	Ballantine and Tanner (2010)
Red mud (bauxite residue)		Filterable	20–98	110–200	Vlahos et al. (1989)
Alum to pasture		Filterable	5–30	150 – >700	McDowell and Norris (2014)
Alum to grazed cropland		Filterable	30	180–340	McDowell and Houlbrooke (2009)
Grass buffer strips	Edge of field	Filterable	0–20	30 – >300	McKergow et al. (2007); Hille et al. (2018)
Sorbents in and near streams		Filterable and Particulate	20	400	McDowell, Hawke, et al. (2007)
Sediment traps		Particulate	10–20	>700	Olarieta et al. (1999)
Dams and water recycling		Filterable and Particulate	50–95	(300) – 700[f]	Barlow et al. (2005)
Advanced effluent ponds		Filterable and Particulate	10–50	>10,000	Sukias et al. (2010); Craggs et al. (2014)
Constructed wetlands		Particulate	−426–77	130 – >700[g]	Vymazal (2007)
Natural seepage wetlands		Particulate	<10%	130 – >700[g]	Hughes et al. (2016)

Note: Costs were taken from McDowell et al. (2013) and McDowell et al. (2017).
[a] The numbers in parentheses represent net profit, not cost.
[b] Profitability depends on existing soil Olsen P concentration; greater savings are possible the more P-enriched the soil is beyond the agronomic optimum.
[c] Profitable at about $40–50/ha assuming the strategy is implemented over 10% of the farm.
[d] Includes adjusting clock timings to decrease outwash <10% of inflow, installation of bunds to prevent outwash, and re-levelling of old borders.
[e] Excludes the cost of the wetland.
[f] The upper bound is only applicable to retention dams combined with water recycling.
[g] The potential for wetlands to act as a source of P renders their upper estimates for cost infinite.

P-rich. Modelling showed that across 14 New Zealand catchments applying strategies to critical source areas improved the cost-effectiveness of mitigations by 6–7 times compared to an untargeted blanket application (McDowell 2014).

The use of these strategies is also influenced by co-benefits. For example, if only focusing on cost-effectiveness, in field management strategies would be conducted before those that rely on mopping up P once it has been lost from the system (e.g. amendments or edge of field strategies). However, strategies such as wetlands are often advocated because they have co-benefits such as decreasing sediment or N losses from the farm and improving biodiversity. Co-benefits have been taken into account in voluntary approaches to implementing strategies such as the Dairying Clean Streams accord and Sustainable Dairying Accord, and proposed regulation such as the mandatory fencing of large streams (Ministry for Primary Industries 2017).

Data for the implementation of the Dairy Clean Streams Accord recorded the uptake of practices such as the: fencing-off of permanent streams (wider than 1 m and deeper than 30 cm), installation of bridges as stock crossings, development of a nutrient budget, and use of good FDE practices preventing ponding and runoff of effluent. The first year of the Accord was 2003. Farmers' self-reported implementation of fencing, stock crossings, nutrient budgets and improved effluent practices in 2003 were 54, 92, 17, and due to inconsistent reporting – 0% of farms, respectively across New Zealand (Ministry of Agriculture and Forestry 2005). By 2007 this was 78%, 98%, 98% and 64%, respectively, and 87%, 99%, 99% and 73%, respectively by 2012 (Ministry for Primary Industries 2013). Subsequent auditing of self-reported data indicated that the quantity and quality of implementation was lower than reported, but that differences in quality when averaged across a region (or nationally) would not likely influence trends over time (Sanson and Baxter 2011). Furthermore, independent statistics from the Regional Authorities showed that the percentage of dairy farms complying with region-specific effluent rules increased, while the percentage of significant non-compliance decreased (Table 8). Independent surveys of 4500 farmers across New Zealand also found an increase in the use of fencing between 2013 and 2017 with 82% and 92% of respondents now indicating that small (one to three order streams; Strahler classification) and large (fourth order and higher) streams were fenced, respectively (Manaaki Whenua: Landcare Research 2018).

Once implemented there is often a lag time before strategies or decreases associated with a change in land use (e.g. factor 1) to become fully effective. Furthermore, the phasing in or inconsistent implementation of strategies across different farms often means that it is difficult to establish what strategy worked at a catchment level. Inconsistencies can be alleviated by planning at a farm and catchment level to place and time the implementation of cost-effective strategies. Several catchment studies have provided advice and recorded the effect of implementing farm-scale strategies at the catchment scale. For instance, for three predominantly dairy-farmed catchments in Taranaki (Waiokura), South Canterbury (Waikakahi) and the West Coast (Inchbonnie), mean P concentrations decreased over a 10-year period by about 15%, 30% and 60%, respectively. In the Waiokura this decrease was attributed to less P fertiliser use (to lower enriched soil Olsen P concentrations) and the use of fencing and good FDE practice (Wilcock, Monaghan, Quinn, et al. 2013). In the Waikakahi the decrease was attributed to decreasing soil Olsen P concentrations and improved irrigation practices (Monaghan et al. 2009). In the Inchbonnie catchment this was attributed to better effluent practices and the use of lower amounts

Table 8. The percentage by region of dairy farms in full compliance (after comma = significant non-compliance) with effluent regulations at the beginning (2003/2004), middle (2007/2008) and end (2011/2012) of the Clean Streams Accord.

Region	2003/2004	2007/2008	2011/2012
Northland	34, n/a[a]	43, 26	38, 27
Auckland	79, <1	73, 7	73, 5
Waikato	43, 16	48, 10	72, 12
Bay of Plenty	n/a	76, 9	67, 16
Taranaki	96, 0.5	96, <1	93, 1
Hawkes Bay	64, 1.6	74, 11	80, 3
Manawatu/Wanganui	85, n/a	78, 22	91, 7
Wellington	89, n/a	53, 28	95, 4
Tasman	9, 9.4	93, 2	94, 2
Marlborough	78, 7.0	75, <1	70, 3
Canterbury	52, 1.0	46, 20	70, 9
Otago	97, n/a	83, 8	94, 4
Southland	81, 6.2	65, 13	45, 12
Weighted national mean	n/a	64, 12	73, 10

Note: Data sourced from the Ministry of Agriculture and Forestry (2005) and the Ministry for Primary Industries (2013).
[a] Data not available.

and less soluble P fertilisers (Wilcock, Monaghan, McDowel, et al. 2013; Wilcock, Monaghan, Quinn, et al. 2013). The Inchbonnie case was also supported by a change in policy (see factor 9).

At a large catchment to regional scale, evidence is emerging of the efficacy of sustained efforts to implement strategies. For instance, Snelder (2018) found a weak but significant positive associations between decreasing suspended sediment concentrations, or greater clarity, and the proportion of catchment area involved in the sustainable land use initiative (SLUI) that developed and implemented farm plans across the Manawatu-Wanganui region (Horizons Regional Council). The decrease in sediment load and increase in water clarity is entirely consistent with the predictions in the efficacy of the implementation of farm plans within the Manawatu catchment (Dymond et al. 2010). A healthy 14–17-year-old poplar will protect 8.4 m^2 of ground from slip and or gully erosion (Hawley and Dymond 1988) and similar results occurred with space-planted eucalypts and willows (Douglas et al. 2013). Dymond et al. (2010) used these and other similar observations to build sediment yield models for farms prior to and after implementing a farm plan. Each plan involved a range of targeted conservation tree plantings including production forest, space-planted poplars to control landslides and large gully erosion, and willows on streambanks to control bank erosion. Dymond et al. (2010) predicted the mean sediment discharge of the Manawatu River would reduce from 3.1 to 1.6 million tonnes per when the soil conservation plantings reached maturity on 500 farms. Horizons Regional Council have recently announced that 'SLUI has completed 683 Farm Plans, with 14 million trees planted covering 500,000 hectares and over 570,000 metres of waterways fenced off' (http://www.horizons.govt.nz/news/improved-water-quality-in-horizons-region-shows-mo). Overall stream and river sediment loads are expected to decrease and cause a concurrent reduction the total particulate-P load carried by the river. However, the P concentration of sediment carried by a river may not decrease by the same amount because the SLUI farm plans focus the farmers' attention on developing the soil fertility on lower slope (<25°) class land (rolling and easy hill) to maintain or increase farm productivity. In addition, the slow movement of sediment through a catchment and the dissolution of P from sediment can delay observable improvements in P concentrations (Rogers et al. 2012; Rickson 2014; Zafar et al. 2017).

These intensively monitored catchments together with data from self-reported surveys, independent auditing or self-reporting and national surveys suggest that more strategies are being used and is a probable cause of decreasing stream P concentrations.

(8) *Better awareness and education of rural professionals*

In 2002, the Fertilizer and Lime Research Centre at Massey University was sponsored by NZ Fertiliser Manufacturers' Research Association to develop a course in Sustainable Nutrient Management in New Zealand Agriculture. The course, which also provides an introduction to nutrient budgeting using the nutrient budgeting software – Overseer (AgResearch 2016), was targeted at rural professionals: fertiliser company field officers, farm consultants and regional council officers. Each year since, between 100 and 180 students have taken this course. In 2005 an Advanced Course in Sustainable Nutrient Management was also added to train students how to undertake farm scenario simulation in Overseer in order to develop farm environment plans. At the end of 2018, 2093 students will have taken the first course (Figure 5) and 856 the advanced course (Figure 6).

Among the 856 rural professionals completing the Advanced Sustainable Nutrient Management course the majority of those were from Canterbury, Waikato, and Manawatu-Wanganui regions (Figure 7). This reflects the size and magnitude of primary production in these regions. Although these courses were not available prior to 2002, it is possible but unclear if the number of 'trained' rural professionals has caused stream and river P concentrations to decrease. However, we can say that there is both greater awareness and capability across New Zealand to create and refresh farm environmental plans, which are seen as a key mechanism in improving the awareness and action required to decrease P losses amongst primary producers. Farm plans are now required by many Regional Authorities (see factor 9) (Beef and Lamb New Zealand 2019).

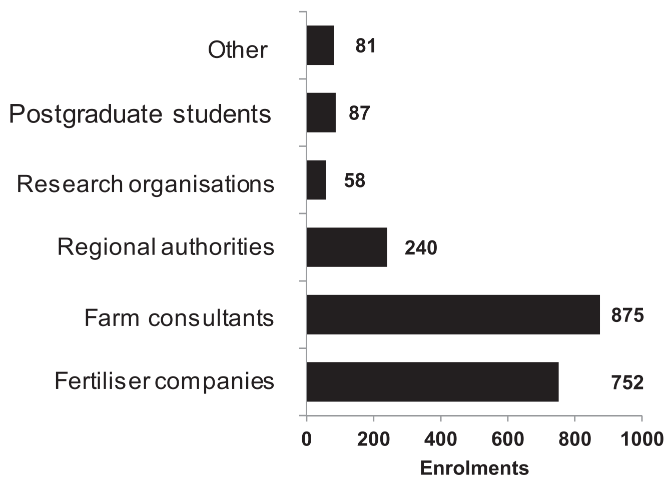

Figure 5. The employment characteristics of the 2093 rural professionals who will have completed a Sustainable Nutrient Management course in New Zealand Agriculture.

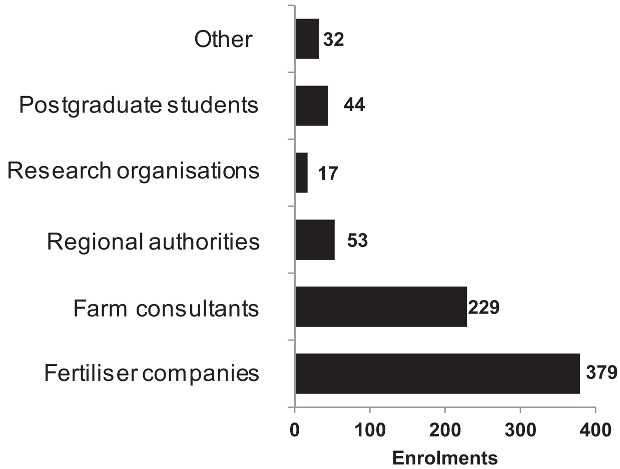

Figure 6. The employment characteristics of the 856 rural professionals who will have completed an Advanced Sustainable Nutrient Management course.

(9) *The use of policy instruments*

The National Policy Statement on Freshwater Management (NPS-FM) was gazetted in 2011, and amended in 2014 and 2017 (Ministry for the Environment 2011, 2014,

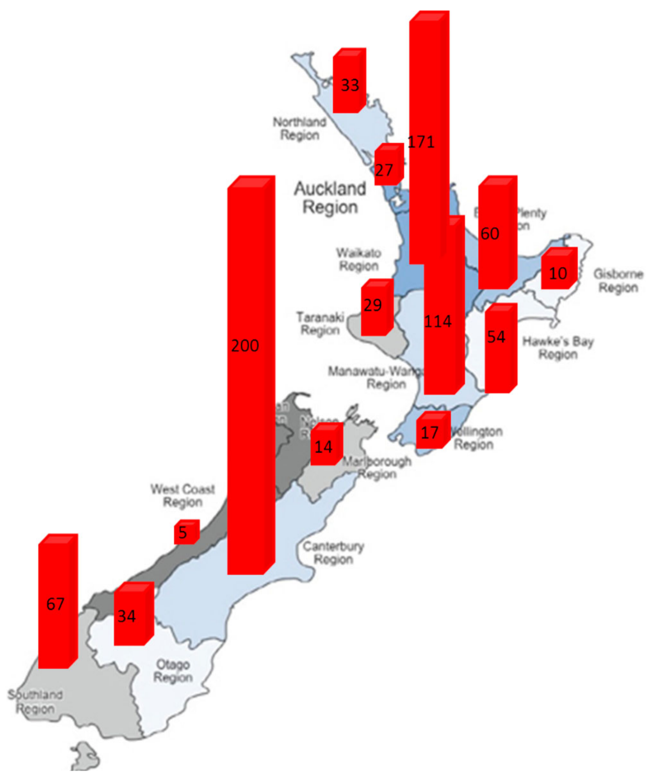

Figure 7. The origin of the 856 rural professionals who will have completed an Advanced Sustainable Nutrient Management course.

2017). It prescribes the minimum standards and some of the steps that Regional Councils have to take to manage water quality. Standards relating specifically to the management of P were added in 2014 and 2017: lakes have numerical targets, while the management of periphyton biomass in streams and rivers requires instream P concentrations to be set.

Prior to the NPS-FM, industry and regional authorities had recognised the role that P plays in water quality and embarked on a mix of voluntary and regulatory approaches to mitigate P loss from land to water. Evidence for this recognition can be seen in the number of New Zealand policy and industry guideline documents recognising P loss from land to water as an issue and identifying the loss of P via critical source areas. As of April 2018, critical source areas were mentioned in 75 policy regulations and industry guideline documents, but not mentioned in documents prior to 2007. This increase is a reflection of a greater awareness of P loss from land on water quality (see factor 8), the maturity of the critical source area concept, and the realisation that targeting mitigation strategies to critical source areas in farm plans improves the cost-effectiveness of mitigations (Liu et al. 2017; Vinten et al. 2017) (Figure 8).

In Australia and New Zealand the time to peak adoption of practices has been measured at between 6 and 22 years: 16.5 years, on average (Botha et al. 2015; Kuehne et al. 2017). Together with catchment processes, the inconsistent implementation of practices means that there is often a lag time to water quality outcomes (Meals et al. 2010; Meter and Basu 2017). The newness of national policy direction in New Zealand such as the NPS-FM and the lead in time allowed for its implementation,[1] infers that few changes will have yet occurred to manage P concentrations in streams and rivers as a result of this instrument. However, some national guidelines were introduced long ago and have

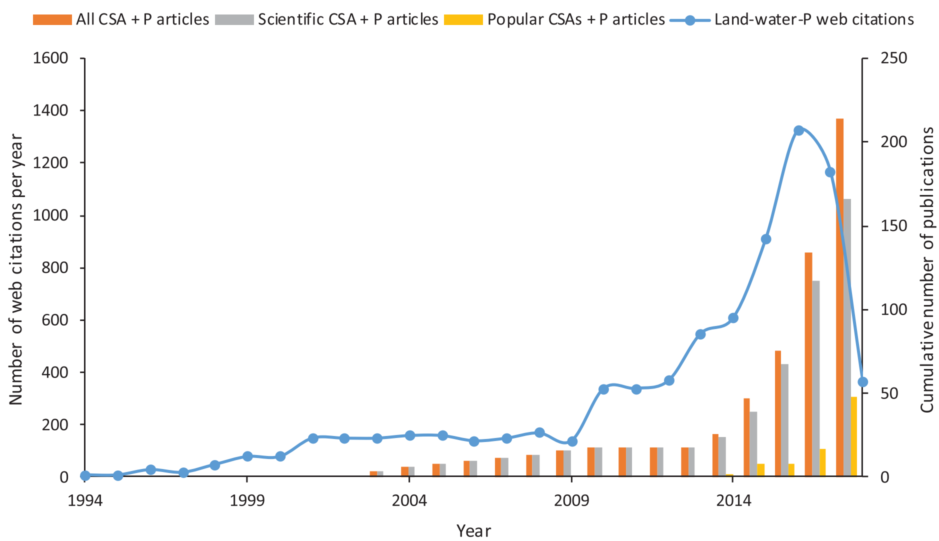

Figure 8. Number of web citations per year and the cumulative number of articles (popular and from scientific journals) on critical source areas (CSAs) and land–water–P interactions over time. Of the 159 scientific documents produced (as of April 2018), 47 were academic papers, 75 were from policy (central or regional), 30 were from industry, while 14 were from non-governmental organisations (Source = Google analytics).

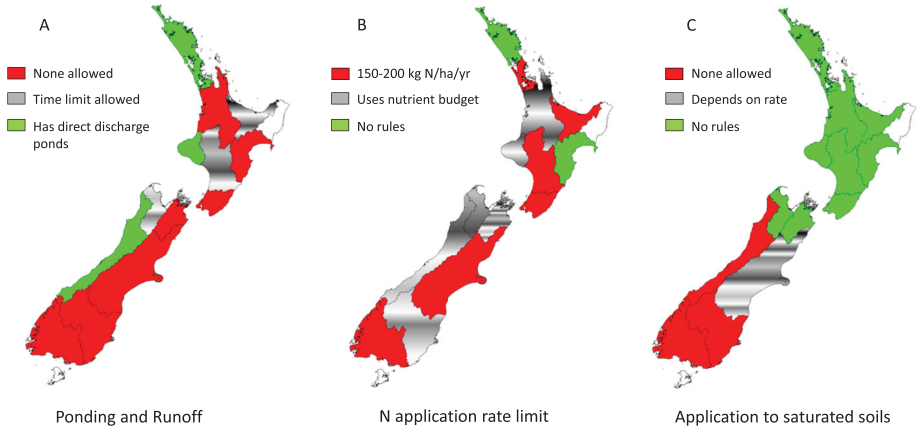

Figure 9. The management of farm dairy effluent by Regional Authorities in New Zealand to decrease N and, by inference, P losses. Management focuses on restricting effluent application to soils to prevent ponding and runoff year-round (A: none, refers to no ponding or runoff allowed at any time; time limit, allows for some ponding and runoff; and direct discharge refers to discharges from effluent settling ponds directly to streams), limiting applications over the year to not exceed a N rate for the effluent block (B: <150–200 kg N ha^{-1} yr^{-1} via direct measurement of the effluent or a nutrient budget), and restricting applications to saturated soils in winter (C: applications are either not allowed, allowed if below a certain rate, or unrestricted). Figure modified from McDowell et al. (2017).

been far reaching, while instances also exist of the early targeting of national guidelines and regional policy. For instance, both guidelines and policy around FDE has been in place in the early 2000s in most regions (McDowell et al. 2017) (Figure 9). These will influence new dairy farms from their date of conversion. Since 1994, the code of practice for the spreading of fertiliser in New Zealand has advised that a buffer zone be left between the application area and waterways and wetlands (Fertiliser Quality Council 2018). Regional policy imposed by the West Coast Regional Council for Lake Brunner in 2010 has restricted the amount and form of P fertiliser and prohibited direct discharge or runoff of FDE (West Coast Regional Council 2010a, 2010b). Such policy has the advantage over industry guidelines that compliance ensures that practices are implemented at a faster rate and more consistently across a catchment (Wang et al. 2012; Kuehne et al. 2017). Data has shown a significant decline in TP concentrations since 2010 (West Coast Regional Council 2017). In Taranaki, Bedford (2017) concluded that a sustained effort to fence 7000 km of waterways in the region since 2000 has helped to significantly decrease median FRP concentrations. The geographic spread and reach of these actions suggest that the increase in policy and guidelines has probably had an effect on decreasing P concentrations.

Conclusions

A semi-quantitative analysis isolated the following causes as unlikely to have caused a decrease in FRP and TP concentrations between 2004 and 2013:

- a decrease in imported P as fertilisers and feed;
- a change to low water-soluble P fertilisers such as RPR;

- a decrease in soil Olsen P concentrations; and
- the greater assimilation of FRP in groundwater and streams and rivers due to greater NO_3 loading.

Although the data were unclear, possible causes of a decrease in P concentrations were:

- land use change resulting in lower erosion and less P applied to sloping land;
- the greater assimilation of P in New Zealand soils – associated with more use of N fertilisers; and
- the better awareness of P as an environmental issue were possible.

Probable causes of decreasing P concentrations were:

- the greater use of strategies to mitigate P loss from land and water;
- the use of guidelines to direct the use of strategies; and
- policy instruments that require better water quality outcomes and, as a result, lower P losses from land to water.

These data support an emphasis on the development and implementation of strategies to mitigate P losses, supported by a mix of voluntary guidelines and regulation to ensure water quality outcomes result. However, improvements in the strength of these findings could be achieved if the analysis was fully quantitative. This analysis would require additional data and privacy issues to be overcome to allow data to be referenced to the same spatial and temporal scales.

Note

1. Regional authorities have until 31 December 2025 to implement the NPS-FM, but an authority may extend the date to 31 December 2030 if it considers that meeting the earlier date would result in lower quality planning; or it would be impracticable for it to complete the implementation of NPS-FM by that date.

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