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Assessing the Yield and Load of Contaminants with Stream Order: Would Policy Requiring Livestock to Be Fenced Out of High-Order Streams Decrease Catchment Contaminant Loads?

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Abstract

Catchment contaminant loads vary with stream order as catchment characteristics influence inputs and in-stream processing. However, the relative influence and policy significance of these characteristics across a number of contaminants and at a national scale is unclear. We modeled the significance of catchment characteristics (e.g., climate, topography, geology, land cover), as captured by a national-scale River Environment Classification (REC) system, and stream order in the estimation of contaminant yields. We used this model to test if potential regulation in New Zealand requiring livestock to be fenced off from large (high)-order streams would substantially decrease catchment contaminant loads. Concentration and flow data for 1998 to 2009 were used to calculate catchment load and yields of nitrogen (N) and phosphorus (P) species, suspended sediment, and *Escherichia coli* at 728 water quality monitoring sites. On average, the yields of all contaminants increased with increasing stream order in catchments dominated by agriculture (generally lowland and pastoral REC land cover classes). Loads from low-order small streams (<1 m wide, 30 cm deep, and in flat catchments dominated by pasture) exempt from potential fencing regulations accounted for an average of 77% of the national load (varying from 73% for total N to 84% for dissolved reactive P). This means that to substantially reduce contaminant losses, other mitigations should be investigated in small streams, particularly where fencing of larger streams has low efficacy.

Core Ideas

- Contaminant yields increased with increasing stream order in catchments dominated by agriculture.
- Fencing off high-order streams from stock misses 77% of national contaminant load from small-order streams.
- Hence, to reduce contaminant losses to small streams, other mitigations are needed.

It is well established that the water quality of streams and rivers is degraded by diffuse agricultural pollution (Carpenter et al., 1998). Catchment limits to improve water quality can be allocated as loads (e.g., kg nitrogen [N] yr^{-1}) or yields (e.g., kg N ha^{-1} yr^{-1}) to individual farms. To help meet these limits, regulators and industry promote farm-scale mitigation strategies that target areas of the farm with the greatest yield (Doody et al., 2012; McDowell et al., 2016a). However, contaminant loads and yields have been shown to vary with increasing catchment and stream size, increasing or decreasing from source to the site of impact downstream depending on flow-paths and associated processes (Bricker et al., 2014). A systematic methodology that describes contaminant losses from headwaters to the catchment outlet is lacking but is important to help policymakers and land managers decide where best to mitigate contaminant inputs or impacts within a catchment (Biggs et al., 2017; Meals, 1996).

Stream orders (Horton-Strahler classification) have been used to characterize stream size and catchment area (Hughes et al., 2011) and to explain variation in contaminant concentrations, loads, and yields (Wigington et al., 1998). When focusing on contaminant concentrations, an almost equal number of studies have found no effect of stream order as those that have. The presence of a stream order effect may be caused by the examination of the effect of stream order in regions where all other significant factors such as land cover are kept equal (Turner et al., 2015). In contrast, studies that show no effect of stream order on contaminant concentrations, loads, and yields often examine one or two stream characteristics, such as stream size, but fail to take into account the (potentially) overriding effects of land cover, hydrology, geology, and land management (Larned et al., 2016). Studies that combine stream order with interactions of other factors such as land cover are rare. In one such study, Buck et al. (2004) found that N and phosphorus (P) concentrations were well predicted in fourth-order streams by the percentage of grazed pasture land cover in the upstream catchment area but

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Abbreviations: DRP, dissolved reactive phosphorus; FDC, flow duration curves; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; REC, River Environment Classification; REML, restricted maximum likelihood; SS, suspended sediment; TN: total N, TP, total P.

that predictions of concentrations were only possible by including land management decisions in second-order streams. We are not aware of a study that has captured and combined multiple characteristics with stream orders across New Zealand.

The River Environment Classification (REC) is a national-scale hierarchical classification of stream segments and upstream catchment characteristics in New Zealand. The REC has been shown to discriminate differences in flow regimes (Snelder et al., 2005), nutrient concentrations (Snelder et al., 2004a), general water quality (Larned et al., 2004), and invertebrate community composition (Snelder et al., 2004b). Our first hypothesis is that by considering the REC, we will be able to model contaminant concentrations, loads, and yields by stream order and that this could be used to test scenarios and inform national policy.

The science used to inform policy for the improvement of water quality often highlights fencing off streams from livestock as a highly effective and quick strategy to mitigate contaminant inputs to streams (McDowell et al., 2017). For example, McDowell (2008) showed the annual yield of surface-derived contaminants such as suspended sediment (SS), phosphorus (P), and the fecal indicator bacteria *Escherichia coli* decreased by up to 90% after fencing off headwater streams from access by farmed red deer. Similarly, James et al. (2007) showed that fencing off streams from dairy cattle could decrease P load in a 4700-ha catchment in southeastern New York by 32%. However, the cost of fencing can be high, ranging from about US\$20 m^{-1} for red deer to about US\$1 m^{-1} for a single wire (Monaghan et al., 2008). If applied to all streams, a farm with many headwaters could therefore face a fencing bill that bankrupts the farm. In recognizing the prohibitive cost of fencing—and that some animals, such as sheep, tend to avoid streams (McColl and Gibson, 1979)—industry guidelines and policy may advise or regulate that only larger, lowland streams be fenced (Dairy Environment Leadership Group, 2013). However, as Alexander et al. (2007) noted for N, most losses occur in the headwaters, implying that a focus on fencing in larger streams may not reduce contaminant loads at regional or national scales.

This paper aims to compare and model contaminant loads according to scale and by catchment characteristics. This model will then be used to determine if guidelines and potential policy that require larger (higher-order) streams to be fenced would be effective in mitigating contaminant loads at a regional or national scale. We use as an example policy the recent advice from the Land and Water Forum (2015), which has been adopted as potential policy in New Zealand (Ministry for the Environment, 2017). This policy states that cattle, deer, and sheep should be fenced out of streams wider than 1 m and deeper than 30 cm at mean annual flow in catchments of mean slope <15 degrees. Although voluntary schemes to fence off streams >1 m wide and 30 cm deep from dairy cattle suggest 90% were fenced by 2013, far fewer large streams on sheep and beef or deer farms are fenced because they are used for stock water, whereas smaller streams are not fenced because there are so many of them that fencing them would be cost prohibitive (Daigneault et al., 2017; Ministry for Primary Industries, 2013). Our second hypothesis is that not fencing narrow, shallow, or sloping streams, hereafter referred to as *exempt* streams, will impair our ability to significantly improve downstream water quality.

Materials and Methods

Data

A database comprising concentrations of SS, nitrate+nitrite nitrogen (NO_x–N), ammoniacal N (NH_4–N), total N (TN), dissolved reactive phosphorus (DRP), total P (TP), and *E. coli* was collated from McDowell et al. (2013) and Larned et al. (2016). The database included 728 sites in New Zealand that have been routinely sampled by regional authorities from as early as the late 1970s. To reduce issues related to changes in water quality analyses and temporal trends, we used data from 1998 to 2009. Data within the database varied widely in reporting formats, reporting conventions, contaminant names, and sampling frequency or flows. To consolidate these data into a uniform structure and minimize the potential for error, we used a modified version of a MS-Access database (Ballantine and Davies-Colley, 2010) and adopted the following filtering conventions for data quality:

1. Sites were only included in the database if there were 50 or more measurements of a contaminant during the period of record, to ensure reasonable coverage of the flow range at the site.
2. Contaminant concentrations less than the indicated detection limit were set at half the detection limit. The percentage of sites where the median concentration was below the stated detection limit was generally <1% except for SS (3.4%), DRP (4.3%), and NH_4–N (17.4%). For contaminant concentrations greater than a censored value, such as *E. coli* (>20,000 most probable number 100 mL^{-1}), the numerical extreme was used.
3. Total N was calculated (where possible) as the sum of NO_x–N plus total Kjeldahl N for regions that did not specifically report this variable.
4. Sites in estuarine waters were omitted to avoid biasing our dataset.

The frequency of sampling varied across the sites represented in the dataset from fortnightly to quarterly. In addition, constraints and objectives associated with the design of regional sampling programs mean that geographical and environmental coverage of the sites is uneven and variable (Fig. 1). The sites in our dataset therefore tended to represent locations where there is a known or anticipated change in water quality due to land use impacts.

We used the New Zealand River Environment Classification (REC) (Snelder and Biggs, 2002) to classify the sites according to the characteristics of the upstream catchment that are strong determinants of their water quality (Table 1). The first four levels of the REC discriminate differences in catchment characteristics in the order of spatially averaged measures of climate, topography, geology, and land cover, respectively. The spatial framework for the REC is a digital representation of the New Zealand river network comprising 576,688 segments (between confluences) and catchments with a mean length of ~700 m that is contained within a geographic information system (GIS).

Geographic coordinates and names were used to assign each water quality monitoring site to a REC class at the first four levels (climate, topography, geology, and land cover) based on the network segment on which it was located (Table 1).

Fig. 1. Location of sampling sites that met our data quality requirements for analysis within New Zealand by region.

Flow Estimation

Contaminant load calculations require stream flow data, both the flow at the time each water quality sample was taken (e.g., mean daily flow) and a representative time series or flow distribution at the site. However, 447 of the 728 water quality monitoring sites did not have flow observations at the time of sampling or with continuous flow gauging records. We used the methods of Booker and Snelder (2012) to estimate flow duration curves (FDC) and mean daily flows on the date corresponding with each water quality sample at each water quality monitoring site.

Load and Yield Calculation

Two methods were used to estimate contaminant yields for each site: regression (viz. rating) and ratio methods. Loads were estimated for each site first and then converted to yields by dividing the loads by the area of the catchment upstream of each water quality monitoring site (kg ha^{-1} yr^{-1}).

The regression method fitted models to the log of concentrations against the log of flow. Following bias correction, to account for back-transformation (Ferguson, 1987), regression model predictions were used to in-fill concentrations at each flow percentile of the FDC. The load associated with each percentile of the FDC was calculated as the product of the corresponding estimated concentration and flow. These individual loads were summed and multiplied by a constant to account for the change of units to produce an annual site load (kg yr^{-1}).

Table 1. Defining characteristics, categories, and membership criteria of selected classes within the New Zealand River Environment Classification at each level.

Level	Defining characteristic (level)	Categories	Notation	Category membership criteria
Level 1	Climate	Warm-extremely wet	WX	Warm: mean annual temperature ≥ 12°C; cool: mean annual temperature < 12°C; extremely wet: mean annual effective precipitation† ≥ 1500 mm; wet: mean annual effective precipitation > 500 and < 1500 mm; dry: mean annual effective precipitation ≤ 500 mm.
		Warm-wet	WW	
		Warm-dry	WD	
		Cool-extremely wet	CX	
		Cool-wet	CW	
		Cool-dry	CD	
Level 2	Topography‡	Glacial-mountain	GM	GM: M and % permanent ice > 1.5%; M: > 50% annual rainfall volume above 1000 m asl; H: 50% rainfall volume between 400 and 1000 m aslL; L: 50% rainfall below 400 m asl; Lk: lake influence index§ > 0.033.
		Mountain	M	
		Hill	H	
		Low-elevation	L	
		Lake	Lk	
Level 3	Geology	Alluvium	Al	Category = the spatially dominant geology category unless combined soft sedimentary geological categories exceed 25% of catchment area, in which case class = SS.
		Hard sedimentary	HS	
		Soft sedimentary	SS	
		Volcanic acidic	VA	
		Volcanic basic	VB	
		Plutonics	P	
		Miscellaneous	M	
Level 4	Land cover	Bare ground	B	Class = the spatially dominant (>50% of catchment area) land-cover category, unless P exceeds 25% of catchment area, in which case class = P or U exceeds 15% of catchment area, in which case class = U.
		Indigenous forest	IF	
		Exotic forest	EF	
		Pastoral	P	
		Scrub	S	
		Urban	U	

† Effective precipitation = annual rainfall – annual potential evapotranspiration.
‡ Called "source of flow" in Snelder and Biggs (2002).
§See Snelder and Biggs (2002) for a description.

The ratio method calculated an annual site load, based on the mean of the product of concentration and flow for days when concentrations were observed (Beale, 1962). This average load was then adjusted by the ratio of the mean flow for all days from the FDC to the mean flow on days when concentrations were observed (Quilbé et al., 2006; Webb and Walling, 1985).

To avoid bias associated with poor representation of very low or high flows, sites were only included where concentrations were available for 90% of the flow range at the site. The regression method was used where the concentration–flow relationship was significant ($P < 0.05$) and the amount of variance explained was >60%; otherwise the ratio method was used, as per Quilbé et al. (2006).

Yield Variation with Stream Order, River Environment Classification Class, and Exclusion by Fencing

For each of the REC classes (climate, topography, geology and land cover), we fitted a restricted maximum likelihood (REML) model (Genstat Committee, 2015) to the log-transformed yields of each contaminant, with stream order as a linear term along with REC class and their interaction; nonlinear order effects were fitted with smoothing splines (Verbyla et al., 1999) on stream order and the interaction of REC class with stream order. We present the significance of differences between REC classes, significance across stream orders and for the interaction between REC class and stream order, and the slope and significance of linear slopes in contaminant yields for stream orders by REC class.

Across REC classes and stream orders, there were 13,230 potential combinations for each contaminant. However, only 2396 occur across the 576,688 stream segments represented in the REC; for example, there are no eighth-order streams of hill topography.

The uncertainty of estimated yields depends on the strength of the relationship between yield and order for each REC class, which is influenced by the amount of data and contributing sites within each class. The REML model does not produce a coefficient of determination that can be used to check of the goodness of fit of the model. However, goodness of fit was assessed using the frequency with which observed yields fall within the mean yield estimated by the model and 95% confidence interval for a class.

Loads from Streams That Do Not Need to Be Fenced

A GIS was used to define the catchment area of each of the 576,688 stream segments represented by the REC. Load predictions were then made for each catchment using the fitted REML models. The methods of Booker (2010) and Jowett (1998) were used within a GIS to isolate those stream segments that were <1 m wide and <30 cm deep and that had a contributing catchment with a mean slope greater than 15 degrees (i.e., exempt streams). The predicted yields for all streams were multiplied by the catchment's contributing area to generate catchment-specific loads for each segment of the REC. The total load (kg yr^{-1}) was calculated for each region and nationally for stream segment catchments meeting the fencing criteria or not for each contaminant for all catchments, and for only those catchments that were dominated by the REC pastoral land-cover class, indicative of intensive land use.

Results

After applying data filtering rules, sufficient data were available to estimate yields for between 243 (SS) and 481 (DRP) sites, depending on the contaminant (Table 2). For TP, NO_x–N, TN, and SS yields, more sites were estimated using the regression than the ratio method, whereas for DRP and *E. coli*, the ratio method was used more frequently (Table 2). A plot (Supplemental Fig. S1) of yields estimated by the two methods across all contaminants yielded a coefficient of determination of 0.98 (regression = $0.94 \times \text{ratio}^{1.0038}$; $P < 0.001$), indicating the outputs from yield calculation methods were, on average, very similar.

The parameters of the REML models, along with their respective 95% confidence intervals, are provided in the Supplemental Table. The fit of the modeled yields to those calculated for each site is indicated by the frequency with which the data fell within the modeled estimate plus or minus the confidence interval. Across all contaminants, 84% of sites fell within the modeled estimate and respective 95% confidence interval, varying from 80% for *E. coli* to 91% for SS (Table 3).

Across all contaminants, there were significant differences among mean yields for climate and geology classes (Table 4). Significant differences among topographical classes were noted for all contaminants except SS. However, for land cover, only DRP and *E. coli* exhibited significant differences among REC classes.

Total P, NH_4–N, and SS yields exhibited significant trends with increasing stream order (Table 5). However, significant trends for other contaminants were dependent on interactions

Table 2. Number and percentage of sites (in parentheses) using the two different yield calculation methods.

Contaminant†	Regression	Ratio	Total
DRP	207 (43)	274 (57)	481
TP	233 (50)	229 (50)	462
NH_4–N	176 (37)	294 (63)	470
NO_x–N	347 (73)	129 (27)	476
TN	328 (72)	131 (28)	459
SS	158 (65)	85 (35)	243
E. coli	119 (27)	329 (73)	448

† DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; SS, suspended sediment; TN, total N; TP, total P.

Table 3. Fit of the restricted maximum likelihood model fitted to each contaminant, expressed as the number and percentage of predicted yields that fell within the mean and 95% confidence interval (CI).

Contaminant†	Sites with yield data	Sites within 95% CI	Percentage of sites within 95% CI
	no.	no.	%
DRP	703	589	84
TP	675	571	85
NH_4–N	687	581	85
NO_x–N	694	611	88
TN	670	587	88
SS	364	332	91
E. coli	655	526	80

† DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; SS, suspended sediment; TN, total N; TP, total P.

Table 4. ***P*** **values for the significance of mean differences in log-transformed contaminant yields between classes within a River Environment Classification level.**

Contaminant†	Climate	Geology	Land cover	Topography
DRP	<0.001	<0.001	0.022	0.050
TP	<0.001	<0.001	0.247	<0.001
NH_4–N	<0.001	<0.001	0.558	0.049
NO_x–N	<0.001	<0.001	0.156	<0.001
TN	<0.001	<0.001	0.667	<0.001
SS	<0.001	<0.001	0.653	0.984
E. coli	<0.001	0.003	<0.001	<0.001

† DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; SS, suspended sediment; TN, total N; TP, total P.

Table 5. ***P*** **values for the significance of a trend in contaminant yields across stream orders and for the interaction of order with river environment classification levels.**

Contaminant†	Order	Order × climate	Order × geology	Order × land cover	Order × topography
DRP	0.414	0.242	<0.001	0.785	0.001
TP	0.012	0.404	<0.001	0.832	<0.001
NH_4–N	0.031	0.675	0.006	0.808	0.040
NO_x–N	0.210	0.024	<0.001	0.513	0.022
TN	0.219	0.640	<0.001	0.993	0.008
SS	<0.001	0.020	<0.001	0.623	0.354
E. coli	0.130	0.925	0.172	0.879	0.104

† DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; SS, suspended sediment; TN, total N; TP, total P.

with hierarchical levels of the REC. For example, both NO_x–N and SS exhibited trends among stream orders of different climate classes. All contaminants but *E. coli* showed trends with order among geology classes. Similarly, all but SS and *E. coli* exhibited trends with stream order among topography classes. However, no trends were observed among order by land-cover classes. An example of this variation in yield with stream order for each REC level is shown in Fig. 2 for NO_x–N.

In addition to assessing the presence or absence of a trend (Table 5), we examined the magnitude and direction of change for the interaction of stream order by climate, geology, topography and land-cover (in log space) by isolating the linear component of each trend (Table 6). This showed that the majority of trends among geology classes were increasing yields with stream order in alluvial streams, and some soft sedimentary and volcanic acid streams, but decreasing yields with increasing order in hard sedimentary streams (Table 6). Significant relationships between yield and order were evenly distributed across climate classes (three contaminants per class), all increasing with stream order. In contrast, all trends in topography were restricted to the lowland class but were consistently increasing with stream order for all contaminants (Table 6).

Predicted yields for all segments of the digital river network were converted into their respective catchment loads. An example of the predicted annual catchment loads of DRP in the Southland region is shown in Fig. 3. In Southland, catchment loads ranged from <1 kg DRP yr^{-1} to >1000 kg DRP yr^{-1}. The proportion of total contaminant loads coming from exempt streams were isolated and compared with those coming from catchments where

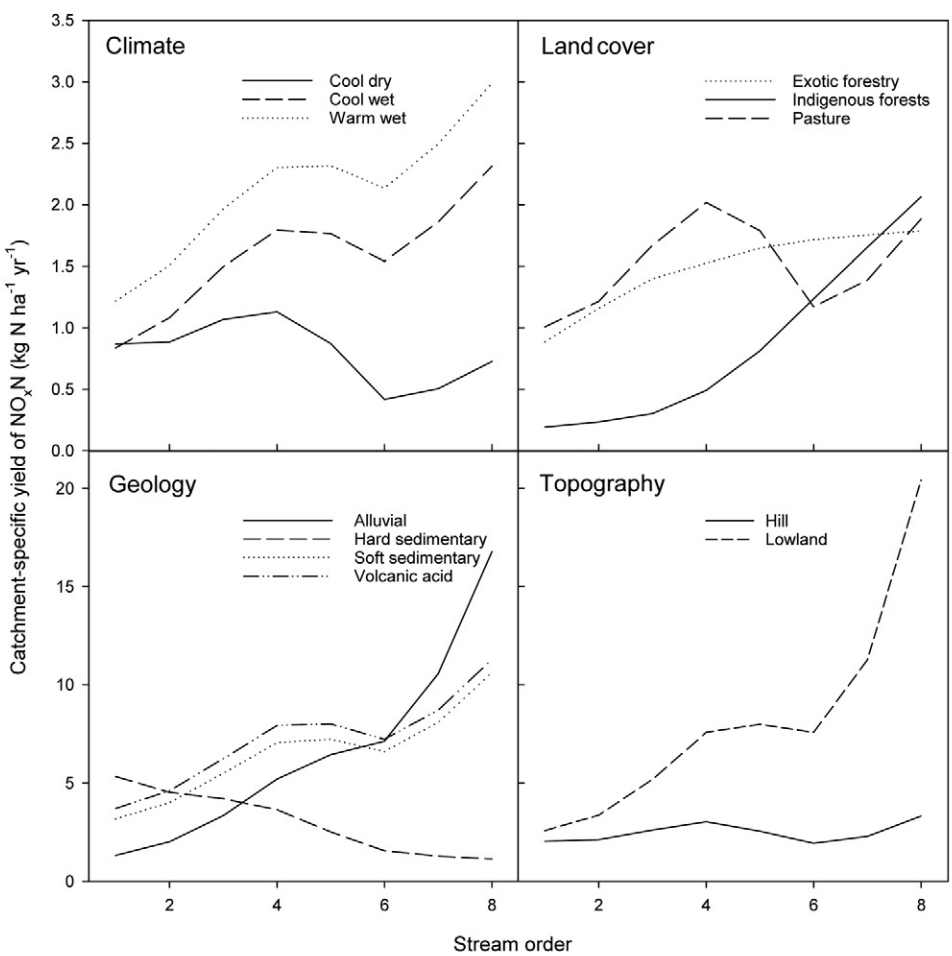

Fig. 2. Variation in mean estimated catchment-specific yield of nitrite + nitrate N for the River Environment Classes: climate, topography, geology, and land cover.

Table 6. Slope (strength), standard error (SE), and significance (bold if $P < 0.05$) of log-transformed slopes for the linear trend in contaminant yields across stream orders by river environment classification class.

Contaminant†		Climate			Geology				Land-cover			Topography	
		Cool dry	Cool wet	Warm wet	Alluvium	Hard sedimentary	Soft sedimentary	Volcanic acid	Exotic forestry	Ingenious forestry	Pasture	Hill	Low-elevation
DRP	Slope	0.15	−0.02	−0.01	**0.32**	**−0.34**	0.02	−0.03	0.18	0.06	0.04	−0.14	**0.19**
	SE	0.08	0.07	0.10	0.08	0.08	0.08	0.07	0.20	0.15	0.07	0.09	0.07
TP	Slope	**0.23**	**0.18**	**0.35**	**0.32**	**−0.31**	0.12	**0.22**	0.12	0.27	**0.20**	0.01	**0.40**
	SE	0.08	0.07	0.11	0.08	0.08	0.09	0.07	0.22	0.16	0.08	0.10	0.09
NH_4–N	Slope	**0.19**	0.16	**0.27**	**0.32**	−0.13	0.12	0.15	0.03	0.16	**0.17**	0.04	**0.30**
	SE	0.09	0.08	0.12	0.10	0.10	0.11	0.09	0.22	0.16	0.08	0.10	0.09
NO_x–N	Slope	−0.06	**0.17**	0.21	**0.34**	**−0.24**	0.15	0.14	0.12	0.28	0.06	0.03	**0.26**
	SE	0.09	0.08	0.11	0.10	0.10	0.11	0.09	0.22	0.17	0.08	0.09	0.08
TN	Slope	0.05	0.12	0.14	**0.35**	**−0.19**	0.09	0.08	0.07	0.09	0.07	0.02	**0.24**
	SE	0.08	0.07	0.10	0.09	0.09	0.09	0.08	0.19	0.15	0.07	0.08	0.07
SS	Slope	0.11	**0.56**	**0.68**	**0.48**	**−0.59**	**0.68**	**0.55**	0.10	0.66	**0.38**	0.27	**0.49**
	SE	0.12	0.14	0.20	0.15	0.22	0.18	0.18	0.49	0.35	0.11	0.18	0.14
E. coli	Slope	**0.27**	0.21	0.22	**0.30**	−0.13	0.04	0.14	0.27	0.20	0.15	0.15	**0.36**
	SE	0.12	0.12	0.16	0.14	0.14	0.15	0.13	0.27	0.19	0.10	0.11	0.09

† DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_x–N, nitrite + nitrate N; SS, suspended sediment; TN, total N; TP, total P.

policy would require fencing across New Zealand (Fig. 4). The mean proportional load coming from catchments requiring fencing was 16% across all contaminants, varying from about 11% for SS to 21% for NO_x–N. By difference, contaminant loads coming from exempt catchments were on average 84% of total load. If only focusing on pastoral land cover (i.e., with grazing animals), the same calculation showed that a lower proportion, amounting to 77% across all contaminants, was coming from exempt catchments; the variation ranged from 73% for DRP and TN to 84% for SS (Fig. 4). Interregional variation was greater still in pasture-dominated catchments, varying from 48% for DRP and TN in the Otago region to 99% for most contaminants in the West Coast region. Agriculturally productive regions such

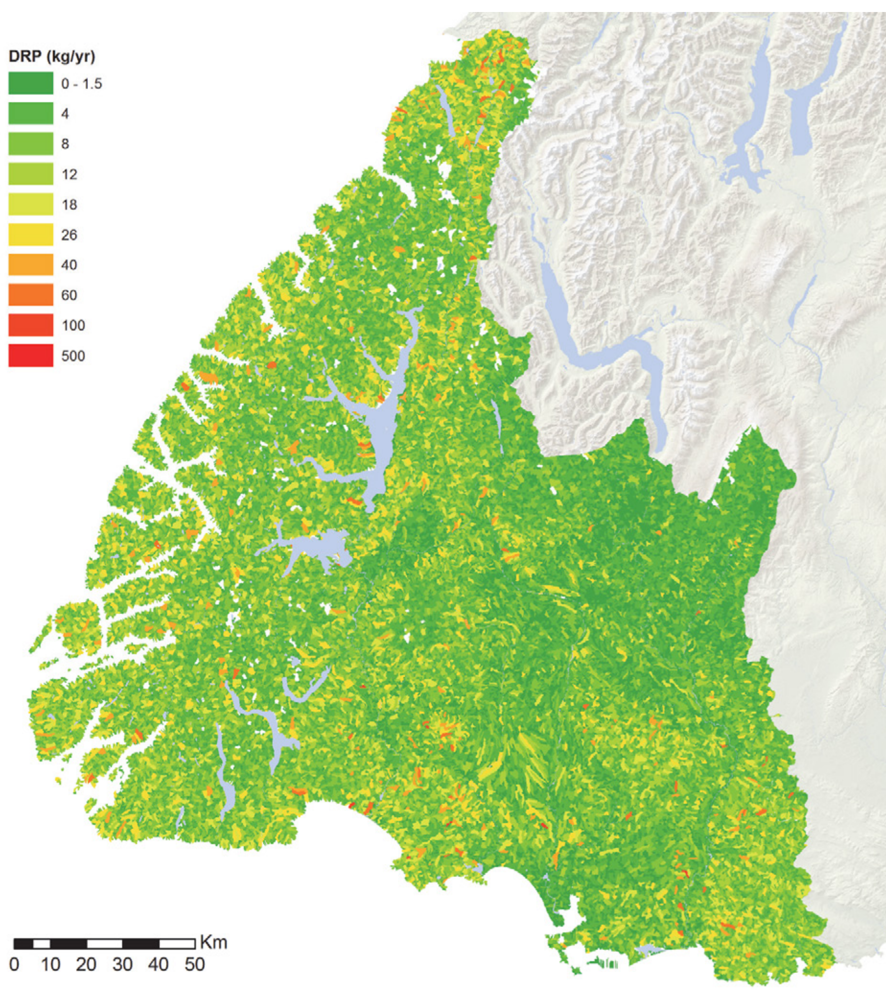

Fig. 3. Estimated total annual load of dissolved reactive P (DRP) (kg yr⁻¹) for all catchments (order 1 through 8) in the Southland region of New Zealand.

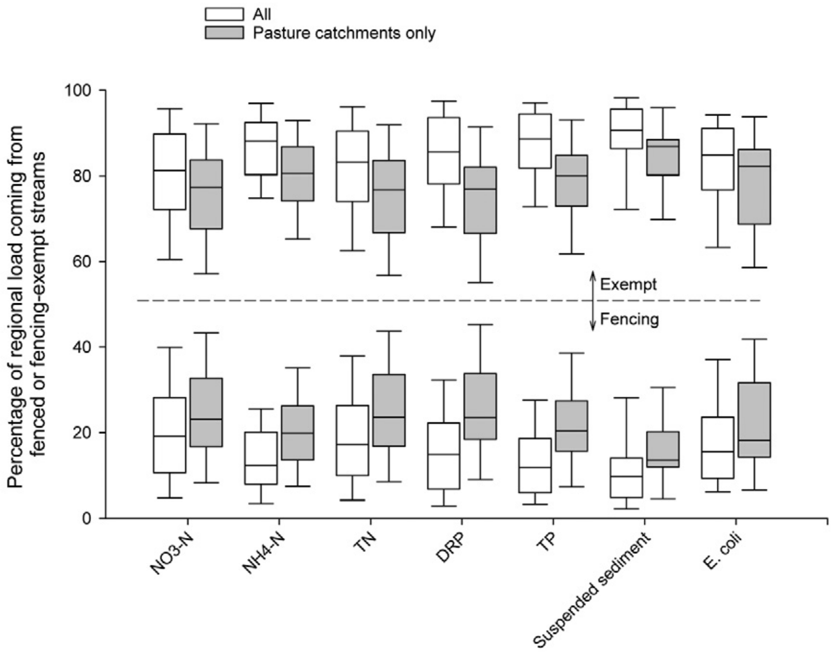

Fig. 4. Box plots showing the percentage mean loads across the 16 regional authorities of New Zealand contributed by fenced and fencing-exempt streams for all land uses and only those under pasture. The 25th and 75th percentiles are the lower and upper ends of the box, with 10th and 90th percentiles as whiskers. DRP, dissolved reactive P; NH_4–N, ammoniacal N; NO_3–N, nitrite + nitrate N; TN, total N; TP, total P.

as Canterbury, Southland, and Hawkes Bay also exhibited large contaminant loads from exempt catchments.

Discussion

Variation of Contaminants with Stream Order

The calculated and modeled contaminant yields in the Supplemental Table and in Fig. 2 are generally low compared with those estimated in other parts of the world. This reflects the large amount of land in New Zealand (41%) that is nonproductive and is held in the conservation estate, in scrubland, or is bare (Anastasiadis et al., 2014). However, comparison of contaminant yields from productive land under pastoral land cover indicates that yields for some contaminants were similar to those generated in grassland-dominant catchments of other well-developed countries (Bilotta et al., 2007; Carpenter et al., 1998; Oliver et al., 2005; Powers et al., 2016).

While trends of contaminant yields varied with stream order, much more contaminant-specific information was gained from significant trends associated with interactions between stream order and REC classes. Among REC classes, trends were most frequently isolated within sites of alluvial geology and lowland topographical REC classes followed by the pastoral land-cover class. Although all of these trends were increasing with stream order, the mechanisms producing these trends will differ among contaminants. Generally, and for brevity, these trends can be characterized as contaminants lost largely via groundwater or runoff pathways that vary according to a range, not one or two, of catchment characteristics.

Nationally, N lost via leaching from top soils is about double that measured in-stream, suggesting a removal (viz. attenuation) of 50% during transport, although that figure is likely to vary spatially (Oehler and Elliott, 2011). However, once lost from top soils, total N and NO_x–N commonly accumulate in deep groundwater and enrich baseflow concentrations downstream (Woodward et al., 2013). This enrichment is driven by a combination of deeper, more N-enriched groundwater that intersects larger rivers downstream (Modica et al., 1997; Puckett, 2004), greater rates of N exchange and removal within the hyporheic zone, biotic activity (e.g., periphyton uptake and denitrification) in smaller streams (Gomez-Velez and Harvey, 2014; Kellogg et al., 2010; Tank et al., 2008), and a greater likelihood of flatter more intensively used and N-leaky land in higher-order catchments (Dodds and Oakes, 2006; Niyogi et al., 2007). Other studies with increasing proportions of intensive land uses showed similar increases for nitrate–N, DRP, and TP with stream order (Rinella and Janet, 1998, Van Nieuwenhuyse and Jones, 1996; Zhou et al., 2012). As supporting evidence for increases, the national dairy herd has doubled between 1997 and 2012 (DairyNZ, 2016), with most of the increase occurring in catchments that would be classified as lowland and with alluvial geology. This reconciles well with the trends shown in Fig. 2. In contrast, some studies have shown decreases with increasing scale where land cover is dominated by forests that take up more N than is produced (Binkley et al., 2004). A decrease was noted for NO_x–N with increasing scale for sites dominated by hard sedimentary geology (Table 5). Such sites tend to dominate high-rainfall conservation areas where the land cover is almost exclusively native and N-hungry *Podocarp* forest.

For all other contaminants, the main pathway of loss is runoff—a combination of surface runoff and subsurface flow—which may include some shallow groundwater (McDowell et al., 2006). Distances and travel times between contaminant sources and the stream channel increase with stream order, resulting in more opportunity for surface-derived contaminants to be attenuated or removed before they enter the stream network (Haygarth et al., 2005; Wood et al., 2005). In theory, runoff-derived contaminant yields should be greater in smaller streams, with a greater proportion of flow as surface runoff than larger streams. However, this does not take into account likely interactions between REC classes and scale, nor the transport of contaminants in fine particles or flocs that are unlikely to settle out under stable flows in larger streams or rivers (Droppo and Ongley, 1994; Stone et al., 1995). For example, higher-order streams tend to have adjacent land that is characterized by lowland topography, which supports more intensive (e.g., pastoral) and diverse land covers than upland catchments (Dodds and Oakes, 2008). Losses of most contaminants increase with land use intensity (Harding et al., 1999), and specifically with the proportion of catchment in pastoral land use (McDowell et al., 2013). Although across all contaminants, no significant trends were noted for land cover, runoff-derived contaminants (TP, NH_4–N and SS) all increased with stream order under pasture (Table 6). Runoff of DRP, TP, and *E. coli* may have been augmented by point source inputs under intensive land use, for example, farm dairy shed wastewater outflows (Edwards et al., 2008).

Implications for Management

Nationally, the proportion of catchments in stream orders 1 to 8 decreased in the following sequence; 51, 23, 13, 7, 3, 2, 1, and <1%. This inevitably means that with the occasional exception (Bricker et al., 2014), headwater streams may account for a larger proportion of the national load than higher-order streams. For example, although calculated using different methods, Alexander et al. (2007) found that the aggregated load of nitrate N from first-order streams was about 42%, decreasing exponentially as stream order increased. We estimated the mean load across regions in New Zealand for NO_x–N from exempt streams (largely headwaters) to be 77% (Fig. 4). The higher percentage probably reflects the inclusion of some second-order streams in the calculation, large areas of upland headwaters in New Zealand, or areas with few streams requiring fencing. For example, while the West Coast region has over 85% pasture land cover, the high average annual rainfall in lowlands (3000–6000 mm, and more falling in uplands) results in few small streams that would need fencing; the resulting contribution of contaminants from West Coast streams requiring fencing is <2%.

In New Zealand, most freshwaters with poor water quality occur in catchments dominated by agricultural land use (Larned et al., 2016). Filtering our analysis to include only catchments dominated by pastoral land cover indicated that a greater proportion of loads for all contaminants were taken into account by streams requiring fencing than when considering all land covers. However, despite removing areas such as the conservation estate from the analysis, large proportions of the total load were from exempt catchments in Canterbury (68%), Southland (71%), Nelson (88%), and Hawkes Bay (72%) regions. These regions have significant downstream rivers that are used for recreation and tourism. Our data suggest that not requiring fencing may significantly delay or reduce the ability to mitigate water quality impairment unless other measures are taken.

In establishing the contribution of headwater loads to higher-order streams, both Alexander et al. (2007) and Lassaletta et al. (2010) caution that neglecting or deemphasizing the contributions of headwater streams in the United States (Clean Water Act) and the European Union (Water Framework Directive) represents a serious impediment to improving water quality at larger scales. Apart from an assumption that headwaters contribute little to catchment loads, the main reason in New Zealand for deemphasizing their role is that it is impractical (e.g., too steep) and too costly to fence them (Ministry for Primary Industries, 2017); consider that the cost to exclude cattle may be about US\$1 m^{-1} but is much greater for red deer at about US\$20 m^{-1} (Monaghan et al., 2008). Nevertheless, high costs may be justified when a large amount of contaminant loading originates from a small area. For instance, McDowell (2007) found that around 90% of catchment loads for TP, SS, and *E. coli* originated from a small area of the catchment (<10%) used by red deer to wallow. Where fencing is prohibitively expensive, a range of less costly strategies is often available that are either contaminant-specific or effective on multiple contaminants (McDowell et al., 2017). These strategies may also be beneficial and act as insurance against the failure of fencing to mitigate contaminant losses. For example, much anecdotal evidence highlights erosion of stream banks and SS loss where fencing is only temporary (McDowell et al., 2016b). Furthermore, contaminant losses can be exacerbated where significant deposition of excreta is associated with stock traffic (e.g., fenceline pacing by red deer) and ephemeral channels carry excreta to the stream (McDowell, 2009).

Previous research on the implementation of on-farm contaminant mitigation strategies has found that the cost-effectiveness of strategies decreases the farther away mitigation occurs from the source (McDowell, 2014). For instance, using the metric of dollar per kilogram P per hectare per year saved, the mitigation of P losses can be nearly cost-neutral by changing fertilizer practices but upward of US\$400 per kg P ha^{-1} yr^{-1} if mitigated via a wetland (McDowell and Nash, 2012). We therefore hypothesize that focusing on contaminant delivery to headwaters, which are not currently required to be fenced (i.e., narrow, shallow, or sloping streams), may be more cost-effective than trying to mitigate delivery or their impact farther downstream. But further work is required to confirm this. Not fencing these streams will likely delay or impair our ability to meet catchment load objectives where fencing of larger, deeper streams in flat areas of the catchment is not effective.

SupplementaL Material

Supplemental data can be found in an online Supplemental Table and Supplemental Fig. S1.

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