BEFORE WAIKATO REGIONAL COUNCIL
HEARINGS PANEL

UNDER	the Resource Management Act 1991 (**RMA**)
IN THE MATTER OF	Proposed Plan Change 1 to the Waikato Regional Plan and Variation 1 to that Proposed Plan Change: Waikato and Waipā River Catchments

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PRIMARY EVIDENCE ON BEHALF OF THE AUCKLAND/WAIKATO & EASTERN REGION FISH AND GAME COUNCILS ("FISH & GAME")

SUBMITTER ID: 74985

Hearing Block 2

Dated: 3 May 2019

Counsel instructed

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1. QUALIFICATIONS AND EXPERIENCE

1.1 My full name is Dr Adam Joshua Daniel.

1.2 I am employed as the Fisheries Manager for the Auckland/Waikato Fish and Game Council at the Hamilton office.

1.3 I have the qualifications and experience set out in the evidence I presented at Block 1 of the Plan Change 1 ('PC1") hearings, dated 15 February 2019.

1.4 I have read the Environment Court's Code of Conduct for Expert Witnesses, and I agree to comply with it. I confirm that the issues addressed in this brief of evidence are within my area of expertise. I have noted (above) that I am employed by the Auckland/Waikato Fish & Game Council.

1.5 I have not omitted to consider material facts known to me that might alter or detract from the opinions expressed. I have specified where my opinion is based on limited or partial information and identified any assumptions, I have made in forming my opinions.

1.6 My opinions rely in part on the Evidence presented by Dr Canning in the Block 1 hearing and Dr Eivers in this hearing Block, for Fish & Game.

2 SUMMARY STATEMENT

2.1 Dr Canning's evidence at Hearing Block 1 set out the range of factors that can affect the ecological community composition in rivers and streams, stating:[1]

> *"All of the biological components of a river food web require the correct habitat, water quantity and water quality in order to maintain healthy populations and functioning ecosystems. A change in a single*

[1] Canning Primary Evidence Block 1 at [3.10].

constituent can have cascading impacts that alter the entire community composition."

2.2 Since Fish & Game's presentation at the Block 1 hearing, staff at Waikato Regional Council and the University of Waikato have published a paper that prioritises the key drivers of invertebrate and fish populations in the Waikato Region, based on sampling of 176 wadable streams in the Region (Pingram *et al.*, 2019). This is extremely valuable as it provides solid data on the key drivers of ecosystem health in the Waikato Region.

2.3 The authors used a robust statistical design to assess the impact of 12 stressors on fish and invertebrate populations. The top four drivers of the 12 assessed, for both invertebrate (QMCI and EPT taxa richness) and fish (Fish QIBI), were:

- Sediment management (identified as the most significant factor);
- Riparian management;
- Instream habitat management; and
- Nutrient management (nitrogen and phosphorus).

2.4 Dr Canning provided evidence on nutrient management in Hearing Block 1. In this evidence I focus on some of the measures necessary to manage the first three factors above, within the context of PC1. I have used Pingram et al. (2019) to frame my interpretation of factors that have led to the degradation of aquatic ecosystems in the Waikato and Waipā catchments. I have also based my evidence on my direct knowledge of the Waikato and Waipā catchments derived from over 10 years working in the catchments.

2.5 The key drivers for riparian habitat management are riparian zone width and the protection of streamside vegetation. Fencing alone will not decrease sediment loads in rivers, without appropriate riparian setbacks (as well as appropriate on-farm practices - covered in the evidence of Dr Eivers).

2.6 Fences located in or adjacent to the riparian zone can create significant problems. Damage to such fences in floods can lead to complaints to

Waikato Regional Council and requests for river works. Such river works can significantly disturb instream habitat for invertebrates and fish - for example the removal of trees can lead to long term loss of woody debris and pools in streams. Adequate riparian buffers in PC 1 would prevent the need for these subsequent management actions, that disturb valuable habitat, or moving of fences later. Adequate buffers will also allow for the establishment of woody vegetation that can supply instream habitat in the future.

2.7 I agree with the recommendations of Dr Eivers in relation to fencing setback distances, for the purpose of Schedule C – Stock Exclusion.

3 KEY FACTS AND OPINIONS

3.1 A recent study of 176 wadable streams in the Waikato Region has ranked the importance of 12 key stressors on invertebrate and fish communities (Pingram et al. 2019; Appendix 1) providing clear guidance on factors limiting ecological health of streams that are within the geographical scope of PC1. The authors concluded that their findings:[2]

1) *"… reveal that between 25 and 50% of mapped target stream length can be considered to be in 'Poor' condition based on biological indices derived from macroinvertebrate and fish community data*."

2) *"… identify that management actions targeted at improving instream habitat quality, particularly reducing fine sediment deposition, when applied across the entire stream network are likely to yield the most widespread improvement in biological condition indices*."

3) *"… shows improving management of sediment, riparian zones and instream habitat is most likely to enhance overall biological condition across the region… ."*

4) *"… highlight the importance of extending policy development beyond a singular focus on water quality if ecosystem health objectives are to be met.".*

[2] Abstract to the Article and under "Conclusions".

3.2 The study compared key stressors to measures of invertebrate and fish health and found that the most important stressors were:

- Sediment management (identified as the most significant factor);
- Riparian management;
- Instream habitat management; and
- Nutrient management (nitrogen and phosphorus).

3.3 It is important to note that Pingram et al. (2019) is focused on the ecological health of wadable streams and does not address lakes, wetlands or large rivers.

Riparian habitat management

3.4 Current forestry (Figure 1 & 2) and grazing practices with no buffers can directly impact streams delivering large amounts of sediment, increasing stream temperatures (Piggott *et al.*, 2012) and directly impact stream health.

3.5 The degradation or loss of riparian vegetation is a key factor in maintaining or restoring ecological health of streams (Sweeney *et al.*, 2004; Holmes *et al.*, 2016) and is known to harm invertebrate and fish communities. Riparian management is directly linked to instream habitat, sediment deposition and water temperature (Quinn et al. 2010). The key drivers for riparian habitat management are riparian zone width and the protection of streamside vegetation. Although a 1 m setback from the top of the bank will improve bank stability, the minimum width of a functional riparian zone is thought to be 5 m (Holmes *et al.*, 2016). A 5 m buffer is a bare minimum, for example even buffers of 15 m have been shown to significantly increase temperatures, deposited sediment and reduce trout production when compared to 30 m buffers (Jones *et al.*, 2006).

Figure 1. Small Waikato stream buried in forestry debris showing direct input of sediment into the small stream (Waikato 2017).

Figure 2. The same Waikato forestry site showing direct input of sediment into a small stream (2017).

3.6 Maintaining upstream riparian habitat is likely more important than adjacent habitat (Piggott *et al.*, 2012; Holmes *et al.*, 2016). I support Dr Eivers evidence that it is important to manage intermittent streams, small wetlands and seeps 'upstream'. Riparian vegetation protection needs to be prioritised from the top down to focus on smaller (Quinn *et al.*, 2010) higher elevation (hill country) habitat if it is to provide lower temperatures, improve habitat and improve water quality in lowland streams. I consider that stock need to be excluded from all streams, and greater gains will be made in terms of improving ecological health if the process is done from the top down (bush to sea). The lack of streamside vegetation also reduces stream width and reduces instream habitat such as wood (Quinn *et al.*, 1997) potentially resulting in poor instream habitat. Therefore, as stated by Dr Eivers[3], setback distance needs to be sufficient to support riparian vegetation to establish (even pasture or native grasses, as a minimum).

3.7 Riparian habitat management alone will not mitigate all of impacts of pastoral farming (Piggott *et al.*, 2012; see section 3.4) or forestry (Jones *et al.*, 2006). Dr Eivers evidence sets out high risk critical source areas that need to be managed on the pasture side of the fence.

Sediment management

3.8 As stated, sediment deposition has been identified as the primary driver of poor ecosystem heath in Waikato streams (Pingram *et al.*, 2019). In my primary evidence for Block 1 I set out the impact of fine sediments on fish and aquatic ecosystems.[4]

3.9 Maintaining stable banks and continual vegetation on stream banks including ephemeral streams and drains is critical for reducing sediment in streams (Figure 3, 4 & 5). Figures 3-5 show examples of the direct impact of agricultural practices observed in the Waikato Region, showing severely degraded riparian vegetation, obvious change in visual clarity

[3] Eivers primary evidence Block 2 [5.19-5.30].
[4] Daniel primary evidence Block 1 at [4.5.2].

and severely damage stream banks resulting from a lack of stock exclusion.

Figure 3. Direct impact of cows in small Waikato stream showing unstable banks, a lack of intact riparian vegetation and significant sediment entering the stream as a result (Ruapuke, Waikato Region 2018).

Figure 4. Ephemeral drain with heavy pugging leading directly into the Mangatutu (Waipā tributary 2017).

Figure 5. Unfenced ephemeral drain with heavy sediment load and bank erosion on a Waikato dry stock farm. A thick layer of deposited sediment is shown on the bed of the drain (2018).

3.10 Similar to instream habitat, sediment management is also heavily influenced by riparian habitat management (Smith and Smith, 2010). Fencing alone will not decrease sediment loads in rivers without on-farm mitigation and appropriate setbacks (Figure 6).

Figure 6. Waipā tributary showing a 1m buffer from the top of the bank with direct runoff from a pasture.

3.11 For Hearing Block 1, I provided a Case Study of the Mangatutu River (upper Waipā catchment) as an example of clearing vegetation on slopes for converting dry stock farm to dairy.[5] Hill country slopes were cleared in approximately 2003 and intensification/conversion to dairy occurred in 2014 on the opposite bank of the river.[6] In the Mangatutu catchment fencing has increased from just over 20% in 2005 to over 70% in 2013 with no reduction in sediment loads (Littler and Berry, 2013). Although bank stability significantly improved over the same period (from under 60% in 2005 to about 90% in 2013) the overall sediment load did not improve. The Mangatutu example highlights need for appropriate setbacks, and highlights the danger of inappropriate farming practices essentially eliminating the improvements made through mitigation such as fencing.

[5] Daniel primary evidence Block 1 at [4.5.9].
[6] Daniel primary evidence Block 1 Figures 1 and 2.

3.12 The need to control erosion on steep slopes is evident throughout the Waikato. It is widely known that slope and tree cover are critical to preventing slips that contribute large quantities of sediment to rivers reducing visual clarity and increasing deposited sediment (Figure 7). Planting native or exotic trees is known to significantly reduce the incidence of slips (Basher, Moores and McLean, 2016) and would reduce sediment loads in most Waikato and Waipā sub catchments.

Figure 7. Steep hill country grazing land with a slip that deposited large a large amount of sediment that required mechanical removal (Waikato stream 2017).

3.13 Stock crossings can contribute sediment from bed disturbance and damage to stream banks in addition to nutrients from defaecation (Wilcock, 2008). Considering that the average dairy herd in the Waikato is 350 cows even occasional crossings can be detrimental to water quality. Similarly, frequent crossings of smaller herds can also cause considerable bed disturbance.

Instream habitat management

3.14 As set out above, instream habitat management was identified as one of the top four drivers of ecosystem health by Pingram *et al.*, 2019. Although this matter is largely outside the scope of PC1, river works in the Waikato Region are often in reaction to complaints of lost fences or races that are located in or adjacent to the riparian zone. Fences located too close to the watercourse can create significant problems. The resulting stream works often degrade important habitat such as gravel bars (Figure 8 & 9), pools, islands and oxbows (Figure 10). These examples are all from the Mangatutu River in just a 7 km stretch of river showing severe degradation of habitat carried out by the Waikato Regional Council. This loss of valuable instream habitat is largely due to the lack of an appropriate riparian buffer. The Waikato Regional Council currently holds emergency works consents that are often used to confine rivers to unrealistically small riparian setbacks.

Figure 8. Left 2000 and right 2013 showing the removal of gravel bars and straitening of the Mangatutu River.

Figure 9. Left 2000 and right 2013 showing the removal of gravel bars and straitening of the Mangatutu River.

Figure 10. Left 2000 and right 2013 showing the removal of gravel bars and an oxbow reclaimed as a paddock on the Mangatutu River.

3.15 The adverse effects of such stream works should not be underestimated. Instream habitat such as boulders, stumps and pools provide necessary habitat for invertebrates and fish (Jones *et al.*, 2006). The lack of instream structures, such as large wood, reduces physical habitat (places for eels to hide) and thwarts the formation of diverse habitat like the pools that are created behind logs. In my opinion there are two primary factors impacting instream habitat in the Waikato and Waipā catchments, the long-term removal of riparian habitat for pastoral farming (covered above) and river works undertaken by the

Waikato Regional Council's Integrated Catchment Management group (ICM) reacting to landowners' requests.

3.16 Although fencing and planting can restore riparian vegetation it will take centuries for native trees from plantings to mature and fall into streams to create habitat (Colley *et al.*, 2010). Many of the riparian zones of Waipā and Waikato streams are covered in willow or poplars that provide valuable instream habitat. Unfortunately, when trees mature and begin to contribute large wood to streams, they are often harvested or removed, preventing any meaningful contribution to instream habitat. This cycle of planting trees such as poplars and willows to protect paddocks and then removing trees when they begin to fall into streams prevents the long term establishment of riparian trees that can contribute to instream habitat.

3.17 The Council's best practice guidelines acknowledge that large woody material "*is an integral part of stream ecosystems*" and "*can provide excellent gradient control preventing the stream bed from degradation*", provide spawning habitat and "shelter during big floods" (Environment Waikato, 2007). As stated, fences too close to a waterbody and the lack of acknowledgement of the riparian zone can have wider ramifications entailing the need for river works, potentially creating significant adverse effects.

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Appendix 1

See attached paper Pingram et al. (2019)

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Improving region-wide ecological condition of wadeable streams: Risk analyses highlight key stressors for policy and management

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ABSTRACT

Unbiased estimates of the current state of target ecosystems and identification of potential causal factors are key to managing stressors over large scales, and for guiding policy and decision makers to set realistic targets and expectations in light of economic pressures. A probability survey design for 176 target wadeable, perennial streams mapped on developed land in the Waikato Region, New Zealand, sampled over three austral summers (2013 to 2015), was used to i) estimate the extent to which “Poor” stressor and biological condition states occur, ii) determine the co-occurrence likelihood of “Poor” biological and environmental condition, and iii) identify and estimate the relative importance of key environmental stressors. The probability survey design also allowed the quantification of uncertainty around mean estimates of extent and risk. These analyses reveal that between 25 and 50% of mapped target stream length can be considered to be in “Poor” condition based on biological indices derived from macroinvertebrate and fish community data. For assessed stressors, Poor condition was estimated for 10 to 50% of the target stream network depending on the stressor. Poor biological condition was likely to co-occur with Poor stressor condition for 10 of the 12 assessed stressors for macroinvertebrate indices, and 5 of the 12 stressors for the fish index. These analyses identify that management actions targeted at improving instream habitat quality, particularly reducing fine sediment deposition, when applied across the entire stream network are likely to yield the most widespread improvement in biological condition indices. Our findings also highlight the importance of extending policy development beyond a singular focus on water quality if ecosystem health objectives are to be met.

1. Introduction

Monitoring of ecological responses to human pressures, and subsequent mitigation interventions, is essential for sustainable environmental management. However identifying causal pathways and mechanisms in multi-stressor settings remains challenging (Leps et al., 2015). Understanding the interplay and co-occurrence between multiple biotic response groups (e.g. macrophyte, macroinvertebrate and fish communities in streams) and environmental stressors (e.g. nutrients, sediment, riparian disturbance) is imperative for integrated catchment management, and for developing achievable national and regional policy targets (Matthaei et al., 2010). Key to determining the relative significance of stressors over broad-scales in target ecosystems is access to unbiased estimates of resource condition and extent across dominant stressor gradients, from pristine through to heavily-modified.

Widely used in human epidemiology, Van Sickle et al. (2006) proposed applying *relative risk* to the association, or co-occurrence, between poor biological condition and high stressor intensity as a tool for identifying which stressors should be the focus of regional- or national-level environmental policy and management. This approach was further developed for biomonitoring application by Van Sickle and Paulsen (2008), who applied the combination of relative extent and relative risk into a single value, *attributable risk*. Attributable risk provides an estimate of the reduction in poor biological condition that could be achieved if values for a given stressor were improved across the target resource to be above a certain condition class threshold. These risk analyses can thus provide important information linking potential causal mechanisms of degradation with changes in the extent of a given biological condition or state in response to management actions. Furthermore, attributable risk analyses should be based on attainable targets for stressor elimination and biological response (Rockhill et al., 1998; Van Sickle and Paulsen, 2008), such as “Poor” and “Not poor” classes.

Biomonitoring of streams has increasingly used sampling network

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designs based on probability theory to provide unbiased estimates of resource condition and extent (Collier and Olsen, 2013; Jiménez-Valencia et al., 2014; Olsen and Peck, 2008; Paulsen et al., 2008), following acknowledgement that previous designs did not adequately describe the overall condition of waterways (Shapiro et al., 2008). A key benefit of this type of monitoring network design is that inferences can be made from a limited number of representative sites with a quantified level of precision, thus making it a highly cost-effective survey approach. Importantly, conclusions drawn from such monitoring networks provide a defensible basis to inform resource allocation, undertake regional state and trend analyses, and develop management options to support the setting of realistic policy goals that achieve environmental outcomes across broad spatial and temporal scales (Peterson et al., 1999).

In the Waikato region of northern New Zealand, biomonitoring of wadeable streams has been conducted since 1994 for a variety of objectives (for more details see Collier, 2005). In 2009, a probability based, spatially balanced survey design was applied to establish a network of wadeable stream monitoring sites on developed land across the region, complemented by an established network of native-forested reference sites. Here, we use this network to inform three of five key questions posed by decision makers - *How big are the problems we have*? *Are they widespread or localized*?, *What are the major causes of the problems*?" (Shapiro et al., 2008). These questions are pertinent in New Zealand, where national legislation directs regional councils (and unitary authorities) to promote the sustainable management of natural and physical resources (i.e. Part II, Resource Management Act, 1991). This aim is further promulgated by the National Policy Statement for Freshwater Management which sets objectives for regional councils to maintain and improve water quality for ecosystem health above nationally set bottom lines (New Zealand Government, 2017).

Regional governments are required to undertake "State of the Environment" monitoring and reporting, primarily to compile and assess information on the condition of the environment, identify pressures on it, and determine what has been and can be done to address key pressures (http://www.qualityplanning.org.nz/ accessed 18 June 2018). Accordingly, we followed the process outlined in Fig. 1 to estimate the relative extent of Poor ecological condition, and identify potential stressor associations (i.e. relative risk) with macroinvertebrate and fish community indices. We then quantified the relative improvements, and explored the magnitude of improvement required, for a given environmental stressor to improve overall biological condition of wadeable streams to a defined "Not-Poor" biological condition (i.e., attributable risk).

2. Methods

2.1. Stream monitoring network

The present study utilises results collected from a probability based sample network of sites on developed land, and sites from a reference network designed to represent an unimpaired condition (native forest), both sampled during 2013–2015. The target population for probabilistic site selection was non-reference (i.e., on developed land), non-tidal, perennial, wadeable streams in the Waikato region (Fig. 2). Sites were selected randomly with a known probability of inclusion using the survey design package spsurvey (Kincaid and Olsen, 2016) in R (R Core Team, 2017). Equal numbers of 1st, 2nd, 3rd and ≥4th order streams were selected (i.e., balanced unequal probability design) using the 'River Environment Classification' (REC) network layer as the sample frame (version 1; Snelder and Biggs, 2002). This survey design ensured an even spread of candidate sites across stream sizes. The REC network layer does not identify all small perennial streams and therefore underestimates headwater network length by an unknown amount. Details of candidate site selection and screening are provided in Collier and Olsen (2013). A total of 491 sites were screened to arrive at 223 target sites, of which 47 were not sampled due to access difficulties, or lack of flow at the time of sampling. As a result, 176 target sites, henceforth 'developed land sites', were sampled (Fig. 2). Surrounding land use at these sites was predominantly pastoral (c.75% of sites), with the remainder made up of native forest or shrubs (13%), exotic forestry (4%), retired pasture (6%), horticulture and urban (each 1%). Up to 60 of these stream reaches were sampled each summer.

Reference sites were located at 27 wadeable stream reaches with upstream catchments in > 80% unmodified native vegetation cover (see Collier et al., 2007; Fig. 2). These reference sites were used to provide a minimally disturbed baseline against which to measure the magnitude of change at other sites and to factor out any regional influences of climatic variation between years (Collier et al., 2007). Each reference site was sampled up to three times during the 3-year study period. Reference site selection was based on achieving a spread of sites across geographic zones (Collier and Hamer, 2012), and dominant stream and environment types across the region based on classifications from the REC and Freshwater Environments of New Zealand (FENZ; Leathwick et al., 2010). These reference sites were used to estimate condition class thresholds for continuous variables (see *Response indices and stressors* section).

2.2. Sampling methods

Sampling was undertaken during the austral summers of 2013–2015 (December–April), and included the simultaneous collection of reach-scale environmental stressor data and biological response information. Each developed land site was visited twice in the year it was sampled; once to undertake fish surveys and once to collect macroinvertebrates and undertake environmental measurements.

Macroinvertebrates were sampled over a 100 m reach using the standard protocols for New Zealand wadeable streams as described by Stark et al. (2001). Briefly, in streams dominated by hard substrates (i.e. those with > 50% benthic substrates made up of gravels, cobbles, boulders or bedrock) up to seven samples from riffles were taken using an aluminium D-net with a 0.5 mm mesh, so that c.0.5–1.0 m^2 of riffle habitat was sampled. In streams where stony substrates were not dominant (i.e. those with > 50% benthic substrates made up of sand, silt, or clay), sampling involved brushing wood, and jabbing the net in and along submerged macrophytes and bankside vegetation at up to 10 locations in run habitats so that an area of c.3 m^2 was sampled. The composite macroinvertebrate sample for each site was preserved in a final concentration of 70% isopropanol and later sorted using a 200-fixed count followed by scan for rare taxa (Protocol P2 of Stark et al., 2001).

Fish surveys were carried out at 144 of the developed land sites using the wadeable stream survey methods of Joy et al. (2013). Briefly, survey reaches were 150 m in length and centred on the same reach used for macroinvertebrate collection with an extra 25 m up- and downstream. In suitable streams all habitats were electro-fished using a NIWA Kainga EFM300 backpack electric fishing machine. The netting protocols of Joy et al. (2013) were used for streams where electric fishing protocols were not suitable (i.e. due to physical and chemical conditions such as high conductivity, or low water clarity). Both techniques provide comparable information for the presence-absence based indices, and a detailed comparison of the two techniques can be found in Joy et al. (2013). Caught fish were identified, counted, and measured (for length) on site before being released. The fewer fish sites than macroinvertebrate sites sampled reflects a small number that were unfishable due to excessive riparian or aquatic plant cover (but which could still be sampled for macroinvertebrates).

2.3. Habitat and physico-chemical conditions

A qualitative habitat assessment (QHA) was conducted at each site using the approach of Collier and Kelly (2005), whereby 8 measures of

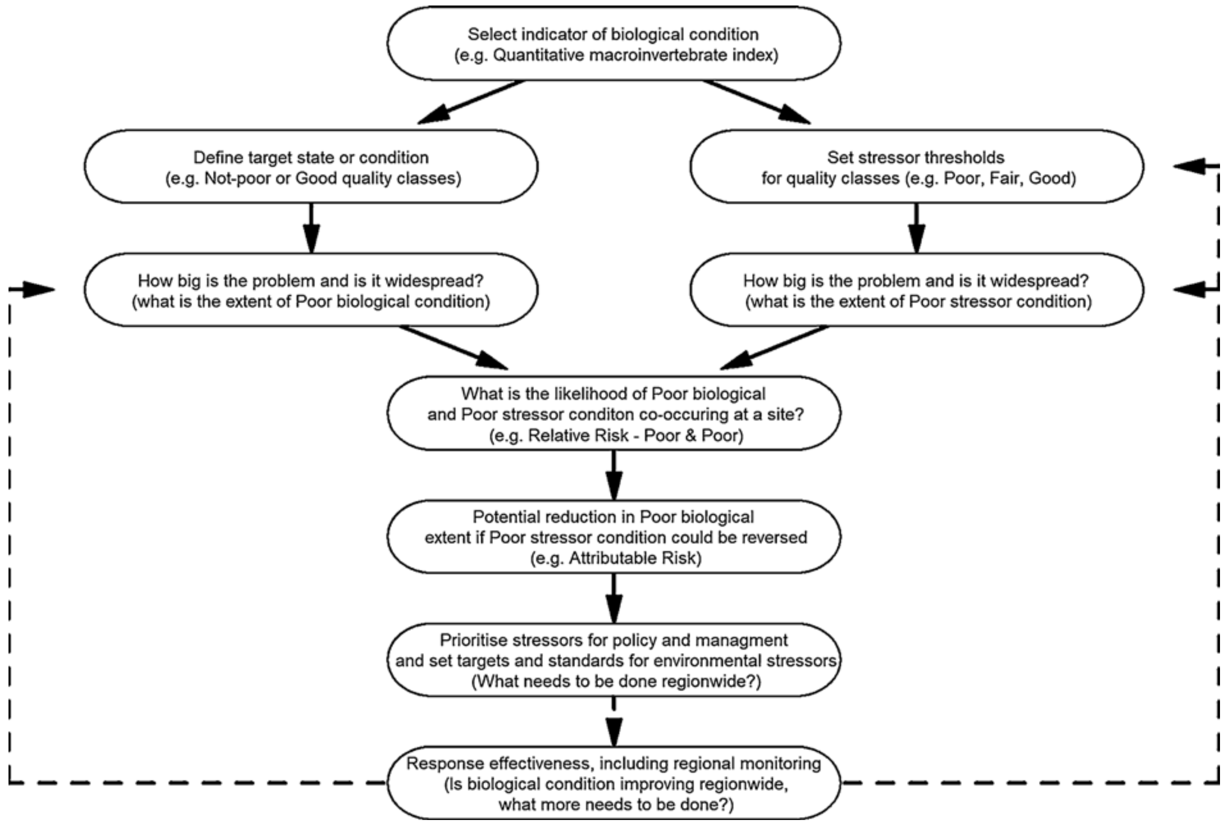

Fig. 1. Flowchart of methodological approach, using collected data. Dashed line indicates re-evaluation steps into the future, and which are not part of the present study.

riparian, bank and channel condition are assessed on a scale of 1 (lowest condition) to 20 (highest condition; see Table S1). Percent bed cover by fine sediment (silt, particle size of < 0.06 mm) was estimated by undertaking a modified Wolman assessment of streambed particles, whereby 100 particles are sampled across five evenly-spaced transects (20 per transect), using the intermediate axis dimension (width) to place the substrate into size divisions. Percent cover of large wood (> 10 cm diameter) was visually estimated for the entire reach.

Water quality samples were taken from just beneath the water surface in flowing sections of water in 1-L polypropylene bottles, and stored on ice until delivery to Hill Laboratories Ltd, Hamilton, for analysis of total nitrogen (TN; mg/L) and total phosphorus (TP; mg/L) (see Table S2 for analytical methods). TN was calculated as the sum of nitrite-nitrate-nitrogen and total Kjeldahl nitrogen.

2.4. Response indices and stressors

Two macroinvertebrate indices for each site, the Quantitative Macroinvertebrate Community Index (QMCI; Stark et al., 2001) and the number of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies; excluding pollution-tolerant Hydroptilidae) taxa (EPT taxa richness). The QMCI was originally proposed to assess organic enrichment in streams, however, it responds to any perturbation where community composition is altered (Boothroyd and Stark, 2000). Tolerance scores used for QMCI calculations were those listed for hard-bottomed streams in Stark and Maxted (2007). For taxa with no score listed, the score at the next taxonomic level up was used, where available, otherwise taxa were excluded from QMCI calculations. EPT taxa richness was used as a measure of taxonomic composition and was highly correlated with calculated presence-absence based Macroinvertebrate Community Index scores ($r^2 = 0.81$, $p < 0.001$). EPT taxa richness was employed as most of these taxa are sensitive to several forms of pollution. One fish index was calculated, the Fish quantile index of biological integrity (Fish Q-IBI; Joy and Death, 2004) modified for the Waikato region (Joy and Henderson, 2007). The Fish Q-IBI is calculated from six sub-measures based on presence-absence data developed to represent relevant environmental factors, including connectivity, habitat, water quality and presence of invasive fish species.

Response and stressor condition classes (Good, Fair, Poor) were assigned based on published categories where they exist, were part of the assessment protocol (e.g. QHA), or alternatively were calculated from data collected at reference sites (see Table 1). Descriptions of published classes are provided in Table S1. Where they existed Excellent condition classes were merged with Good to form a single grouping for the purpose of analysis. Condition classes for continuous variables (e.g. silt), were derived from reference sites using the distribution of values across sites (see Stoddard et al., 2006; Van Sickle et al., 2006 for further details), such that (i) where higher values were indicative of better condition (e.g., cover by large wood) those $\leq 1^{st}$-percentile of reference sites were classed as in Poor condition, those $> 25^{th}$-percentile as Good, and values in between as Fair, and (ii) where higher values were indicative of poorer condition (e.g. TN), the Poor and Good condition class thresholds were set at the 99^{th}- and 75^{th}-percentiles, respectively, and as before values in between were classed as Fair.

2.5. Extent and risk estimates

All extent and risk estimates were calculated using the R software package spsurvey (Kincaid and Olsen, 2016). Standard error and confidence interval estimates were based on the local neighbourhood variance estimation method described by Stevens and Olsen (2003). *"Relative extent"* is the proportion of wadeable stream length on developed land in a given biological or stressor condition class. For the purpose of this analysis, we focussed on stressors measured at a reach scale, and that are actively the focus of management and policy

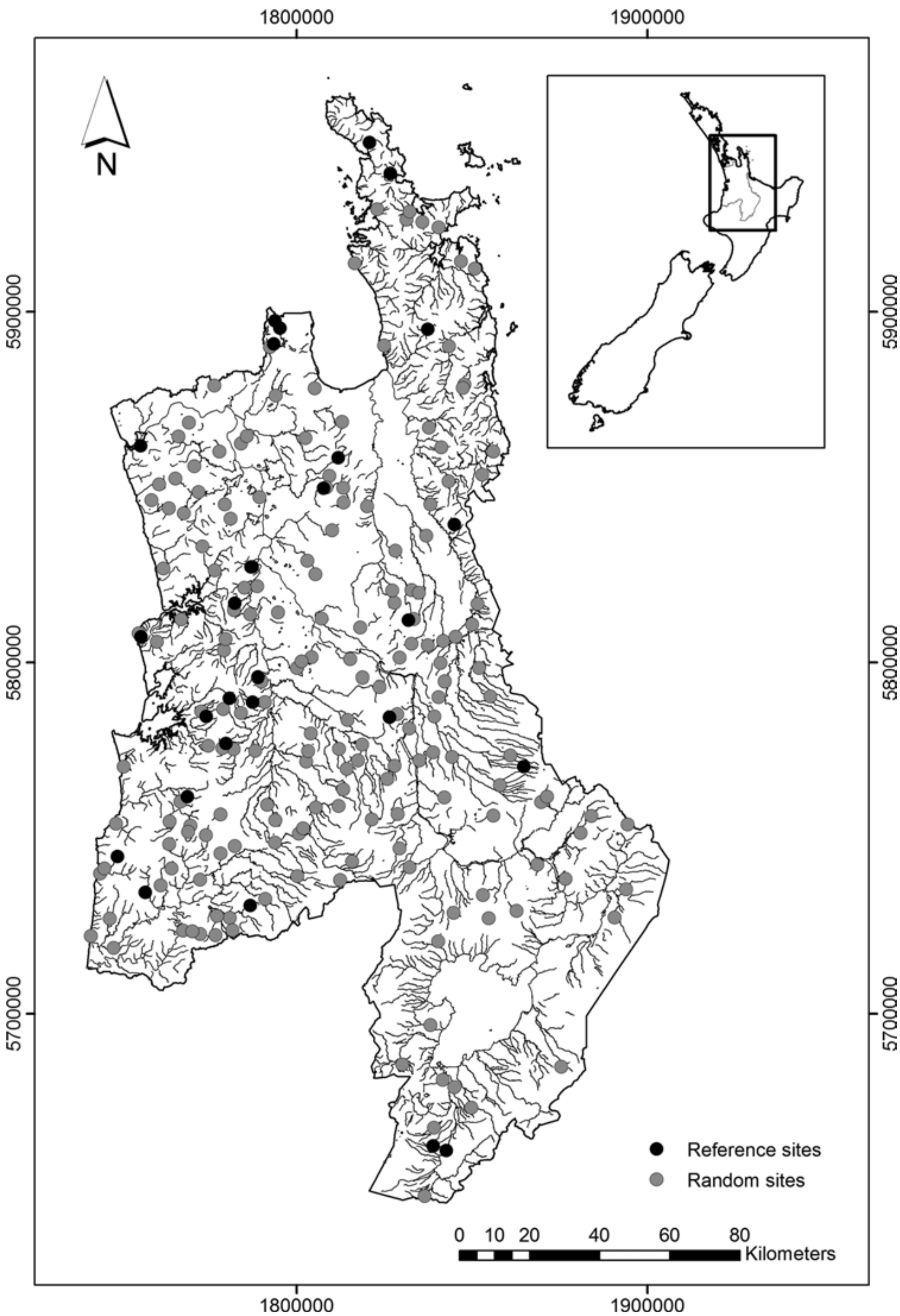

Fig. 2. Map of Waikato region, showing distribution of randomly selected sites on developed land (grey dots), and reference sites (black dots; > 80% native forest catchment cover).

development. "*Relative risk*" is an estimate of the likelihood of a Poor biological condition class co-occurring with Poor condition for a given stressor across a region. A relative risk score of more than 1 indicates that a Poor biological condition is more likely than not to co-occur with a given Poor stressor condition, and the higher the value the more likely this is to be the case. For example, a relative risk value of 2 for riparian condition would indicate that a Poor QMCI score is 2 times more likely to be found in streams with Poor riparian condition. "*Attributable risk*" combines both the extent and relative risk of a given stressor into a single risk value, providing an estimate of the reduction in Poor biological condition that could be achieved if a stressor was to be made Not poor across the entire region (i.e. made Fair or Good in this study; and assuming causality and reversibility after Van Sickle and Paulsen, 2008). An additional attributable risk analysis was performed where the Fair condition class was merged with Poor to form a single grouping. This exercise was intended to explore a) whether further reductions in Poor biological condition could be anticipated by improving region-wide condition of a given stressor to a Good condition, and b) the effect of changing quality class thresholds on the strength of association between Poor condition classes of stressors and biological indicators.

Van Sickle and Paulsen (2008) advocated combining attributable risk estimates for stressors that were either closely correlated, represent closely linked physical, chemical or biological processes that affect the

Table 1
Threshold values, and Poor quality class descriptions where available, for biotic indices and environmental stressors.

Assessment	Management group	Measure	Good	Fair	Poor	Poor quality class description
Biological response indices						
	N/A	Quantitative macroinvertebrate community index[1]	≥5.00	4 – 4.99	< 4	Probable severe pollution
	N/A	Ephemeroptera, Plecoptera and Trichoptera taxa richness[2*]	≥ 9	4 – 8	< 4	Not provided
	N/A	Fish quantile index of biological integrity[3]	≥ 36	27 –35	< 27	Site is impacted or migratory access almost non-existent.
Stressor variables						
Qualitative habitat assessment[4]	Instream habitat management	Abundance and diversity of habitat	≥ 11	6 – 10	1 – 5	< 10% substrate favourable for invertebrate colonisation. Fish cover rare or absent. Substrate unstable or lacking. Stable habitats lacking or limited to macrophytes
	Sediment management	Bank stability	≥ 11	6 – 10	1 – 5	Unstable. Many eroded areas. 60-100% of bank has erosional scars
	Instream habitat management	Channel alteration	≥ 11	6 – 10	1 – 5	Banks shored with gabion or cement. > 80% of the stream reach channelised and disrupted. Instream habitat altered or absent.
	Instream habitat management	Frequency of riffles and channel sinuosity	≥ 11	6 – 10	1 – 5	Generally flat water, shallow riffles. Channel straight. Poor habitat
	Riparian management	Riparian zone width	≥ 11	6 – 10	1 – 5	Breaks frequent. Human activity obvious.
	Sediment management	Sediment deposition	≥ 11	6 – 10	1 – 5	Heavy deposits of fine material. Increased bar development. > 80% of the bottom changing frequently and pools almost absent due to sediment deposition
	Riparian management	Vegetative protection	≥ 11	6 – 10	1 – 5	Bank surfaces covered by grasses and shrubs. Disruption of streambank vegetation very high. Grass heavily grazed. Significant stock damage to the bank.
	Instream habitat management	Velocity, depth and pool regimes	≥ 11	6 – 10	1 – 5	Dominated by 1 velocity/depth regime. Usually slow/deep. Majority of pools small/shallow.
Water quality[5]	Nutrient management	TN (mg/L)	≤ 0.314	0.315 – 0.662	> 0.662	> 99th percentile of reference sites
	Nutrient management	TP (mg/L)	≤ 0.046	0.047 – 0.108	> 0.108	> 99th percentile of reference sites
Substrate[5]	Sediment management	Silt (% cover)	≤ 2	3 – 18	> 18	> 99th percentile of reference sites
Large wood[5]	Riparian management	Large wood (% cover)	> 1	1	0	< 1st percentile of reference sites

Sources: [1] Stark and Maxted (2007), [2] Collier and Hamer (2013), [3] Joy and Henderson (2007), [4] Collier and Kelly (2005), [5] reference site percentiles. *excluding Hydroptilidae. N/A = not applicable. See Table S1 for detailed descriptions of Poor to Excellent quality classes of those derived from cited sources.

response variable, and/or which would be likely to be managed or approached together more so than with other groups of stressors. Therefore, a combined stressor analysis was also performed whereby if any one of the grouped stressors was in Poor condition then a site is considered Poor for that group. Using this approach, we combined mechanistically-linked stressors into the following management groupings: (i) nutrients (TN and TP), (ii) riparian vegetation (Vegetation protection, Riparian zone width and Large wood), (iii) sediment (Bank stability, Overall sediment deposition, Cover by silt), and (iv) instream habitat (two measures of hydraulic heterogeneity (Frequency of riffles/channel sinuosity and Velocity, depth and pool regimes), and Abundance and diversity of habitat). Pearson correlation coefficients indicated that, in general, stressors within groups were better correlated with each other than between groups (Fig. S1). For risk analyses, only stressors with a lower 95% confidence interval > 1 for relative risk, and > 1% for attributable risk were considered reliable as values less than these suggest only weak associations at best (after Van Sickle et al., 2006).

3. Results

3.1. Average biophysical condition

Mean values for QMCI, EPT taxa richness and fish community QIBI for wadeable streams on developed land in the Waikato region were indicative overall of" Fair" biological condition (mean QMCI 4.4, mean EPT richness 7.3; mean Fish-QIBI 32.0; Table S3). Mean QHA stressor variables generally showed a similar pattern to biological response indices (in terms of quality class), being generally in a Fair condition class. Mean physicochemical measures were Poor to Fair: TN (0.99 mg/L), TP (0.15 mg/L), Silt cover (31.7%), except Large wood which was classed as Good (1.5% bed cover).

3.2. Relative extent estimates

Wadeable streams on developed land are estimated to account for c. 15,000 km of the region-wide river network (represented by the REC), equivalent to c.41% of the region's river network (the remaining being made up of a mixture of non-target waterways, including reference condition, tidal, intermittent, and non-wadeable rivers, see Table S4 for further details). Extent estimates for biological condition indices indicated that Poor ecological health occurred at c.49% of target stream length based on the QMCI (< 4), and 34% of stream length for EPT taxa richness (< 4 taxa; Fig. 3A). Around 23% of the region's wadeable stream length on developed land was likely to have a fish community considered to be in a Poor condition (Fish-QIBI < 27; Fig. 3A). For all three biological indices, c. 29–47% of target stream length was estimated to be in Good condition (Fig. 3A).

Less than half of the target network length was estimated to have total nutrient concentrations higher than the 99th-percentile of that measured from reference sites (i.e. Poor) under base flow conditions (TN 38%, TP 21%; Fig. 3B). Around 46% of the network was estimated to have deposited fine silt at levels above 99th percentile observed at reference sites (i.e. > 18%; Table 1, Fig. 3B), while large wood was expected to be completely absent from around 48% of target stream length.

Qualitative scores of riparian vegetation were estimated to be indicative of obvious human activity, disturbed vegetation or grazing, and potential stock damage to the bank (i.e. Poor condition; see Fig. 3B) across approximately one third of target stream length (or 3000–4000 km). There was also likely to be a high degree of Bank erosion and Sediment deposition for around 17% and 36% of stream length, respectively, based on the QHA. Similarly, Habitat abundance and diversity was expected to be largely limited to macrophytes with other stable substrates being largely absent (Poor) across around 31% of the developed land stream network, and hydraulic heterogeneity limited to homogenous channels (Poor) across around 25% of stream length.

3.3. Relative risk

Mean estimated relative risk (likelihood of a stressor and a biological response both being in a Poor quality class) for QMCI ranged from 1.3 for Bank stability (i.e. Poor quality classes 1.3 times more likely to co-occur than not) to 2.3 for Frequency of riffles/channel sinuosity (i.e. Poor quality classes 2.3 times more likely to co-occur than not; Fig. 4). All stressors, except for Bank stability and TN, were likely to co-occur with Poor QMCI scores when in Poor condition (i.e. had a lower 95% confidence interval > 1; Fig. 4). For EPT taxa richness, the relative risk values of the investigated stressors ranged from 1.5 (Riparian zone width) to 9.5 (Silt cover; Fig. 4). As with QMCI, all but two variables (Bank stability, Riparian zone width) were more likely than not to co-occur with Poor scores when in Poor condition.

Relative risks for Fish QIBI from investigated reach-scale stressors ranged from 0.6 (Vegetative protection) to 3.8 (TP), however, unlike invertebrate indices relative risk estimates were only compelling for around half the investigated stressors as most had lower 95% confidence intervals < 1 (Fig. 4). Stressors with lower 95% confidence intervals > 1 were TP, Velocity/depth/pool regimes, Sediment deposition, Abundance and diversity of habitat, and % Silt cover. In summary, biological condition was more likely to be Poor where an individual stressor condition was also assessed as Poor for 10 of the 12 environmental stressors for each of the macroinvertebrate indices, and 5 of the 12 environmental stressors for the fish QIBI.

3.4. Attributable risk

For QMCI and EPT taxa richness, the mean estimated reduction in Poor regional extent, should a given stressor be improved to a Fair or Good condition class across the entire region (attributable risk), was lowest for Bank stability (5% and 9%, respectively) and highest for deposited silt cover (28% and 80%, respectively; Fig. 5). For invertebrate indices, other stressors that if improved to a Fair or Good condition would reduce the extent of the Poor biological condition by more than 20%, included Abundance and diversity of habitat, both measures of hydrological heterogeneity, Vegetation protection, and % Large wood. Sediment deposition and nutrients (TN, TP) were also significant for EPT taxa richness. Improving stressor condition to Good would likely increase the estimated improvement in region-wide QMCI quality classes, most notably for Vegetative protection, Abundance and diversity of habitat and Bank stability. Similarly, for EPT taxa Vegetative protection, Abundance and diversity of habitat, aspects of hydrological heterogeneity, Sediment deposition, Bank stability, and TN were identified as compelling factors for additional management (Fig. 5).

Mean estimated attributable risk for Fish-QIBI was up to 37% for % Silt cover, however, several stressors had estimates < 1%, reflecting variable associations. While a number of stressors had mean attributable risk estimates of > 20%, only TP and Velocity/depth/pool regimes had lower confidence intervals of > 1% (Fig. 5). Improving stressor condition to Good appeared to yield little additional improvement in attributable risk estimates for Fish-QIBI (Fig. 5).

3.5. Combined stressor attributable risk estimates

The mean attributable risk estimates of individual stressors summed to greater than 100% (indicating that relationships are likely to exist between individual stressors; Fig. 5). Scenarios of combined related stressors, as recommended by Van Sickle and Paulsen (2008), reduced the sum of mean attributable risk estimates for QMCI, but these remained high for EPT taxa richness (Fig. 5). It was estimated that improving sediment, riparian and instream habitat management groups to a Not Poor condition could reduce the extent of Poor QMCI scores by

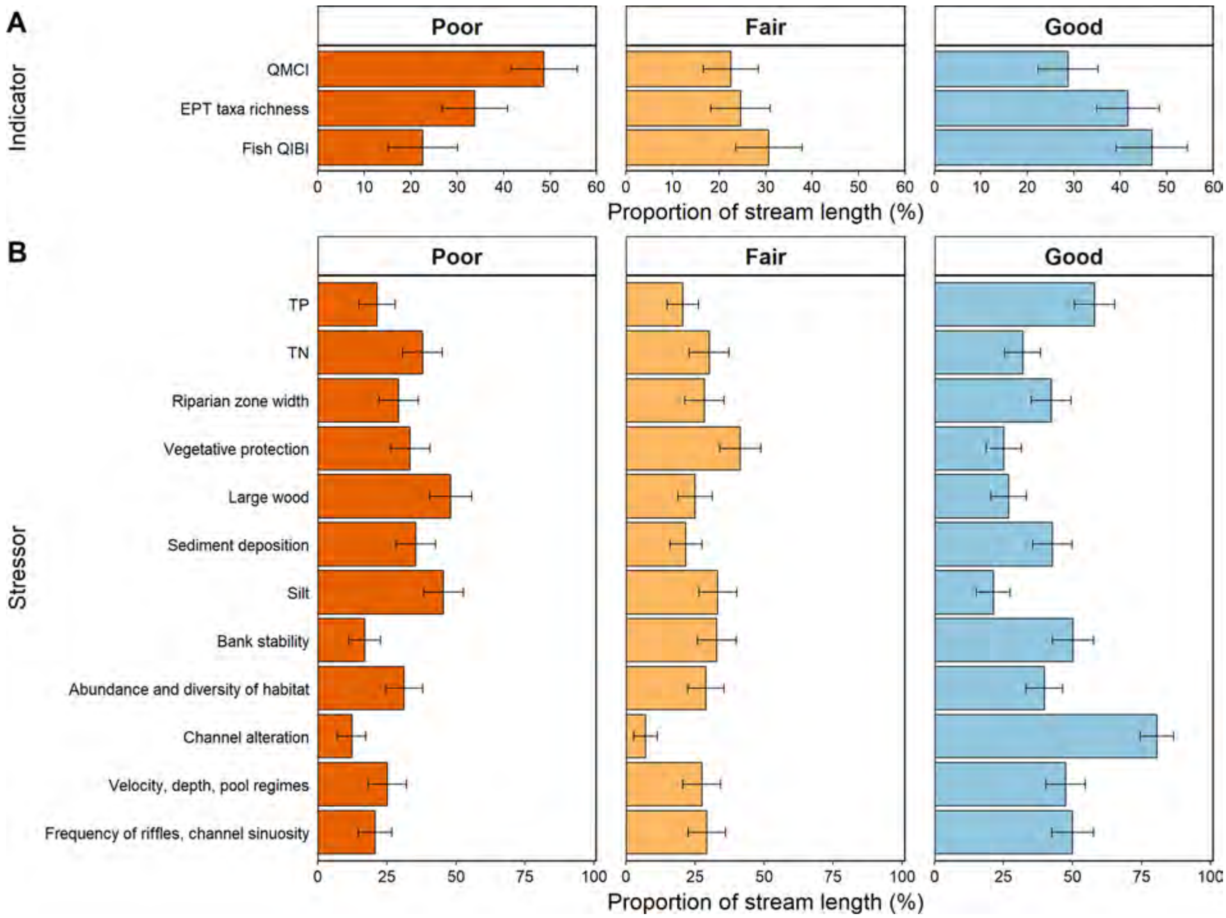

Fig. 3. Mean percentage of wadeable stream length on developed land estimated to be in each of the three quality classes for: (A) the Quantitative Macroinvertebrate Community Index (QMCI), EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa richness (excluding Hydroptilidae), and the Fish quantile index of biological integrity (Fish QIBI), and (B) assessed environmental stressors in the Waikato region. Error bars represent 95% confidence intervals.

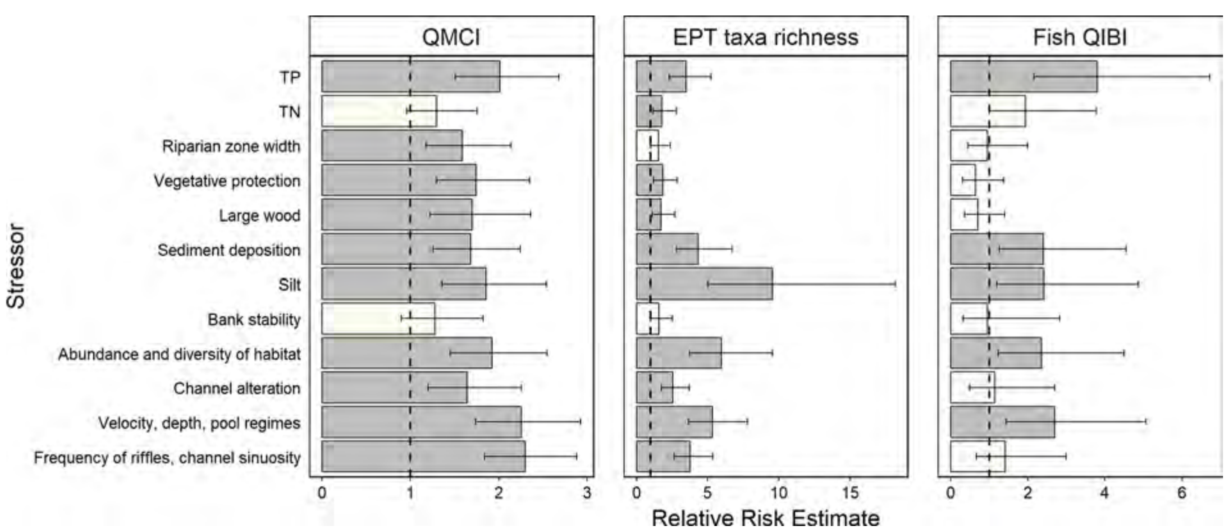

Fig. 4. Mean relative risk estimates against measured stressors (likelihood of co-occurrence) for the Quantitative Macroinvertebrate Community Index (QMCI), EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa richness (excluding Hydroptilidae), and the Fish quantile index of biological integrity (Fish QIBI). Dashed lines indicate a relative risk estimate of 1 (values of > 1 are more likely to co-occur than not). Error bars represent 95% confidence intervals. Grey bars indicate a relative risk estimate with a lower 95% confidence interval of > 1 and white bars those with a lower 95% confidence interval of < 1.

around a third each, each equivalent to c.2600–2800 km of the stream network (< 1000 km for nutrient management; Table 2). For EPT taxa richness, the attributable risk values for combined management of sediments, instream habitat and nutrients yielded potential mean reductions in Poor biological condition of 85%, 83% and 27%, respectively. Addressing sediment and habitat issues across the entire region, could therefore markedly reduce the extent of Poor EPT taxa richness by a length of > 4000 km. Only nutrient and instream habitat

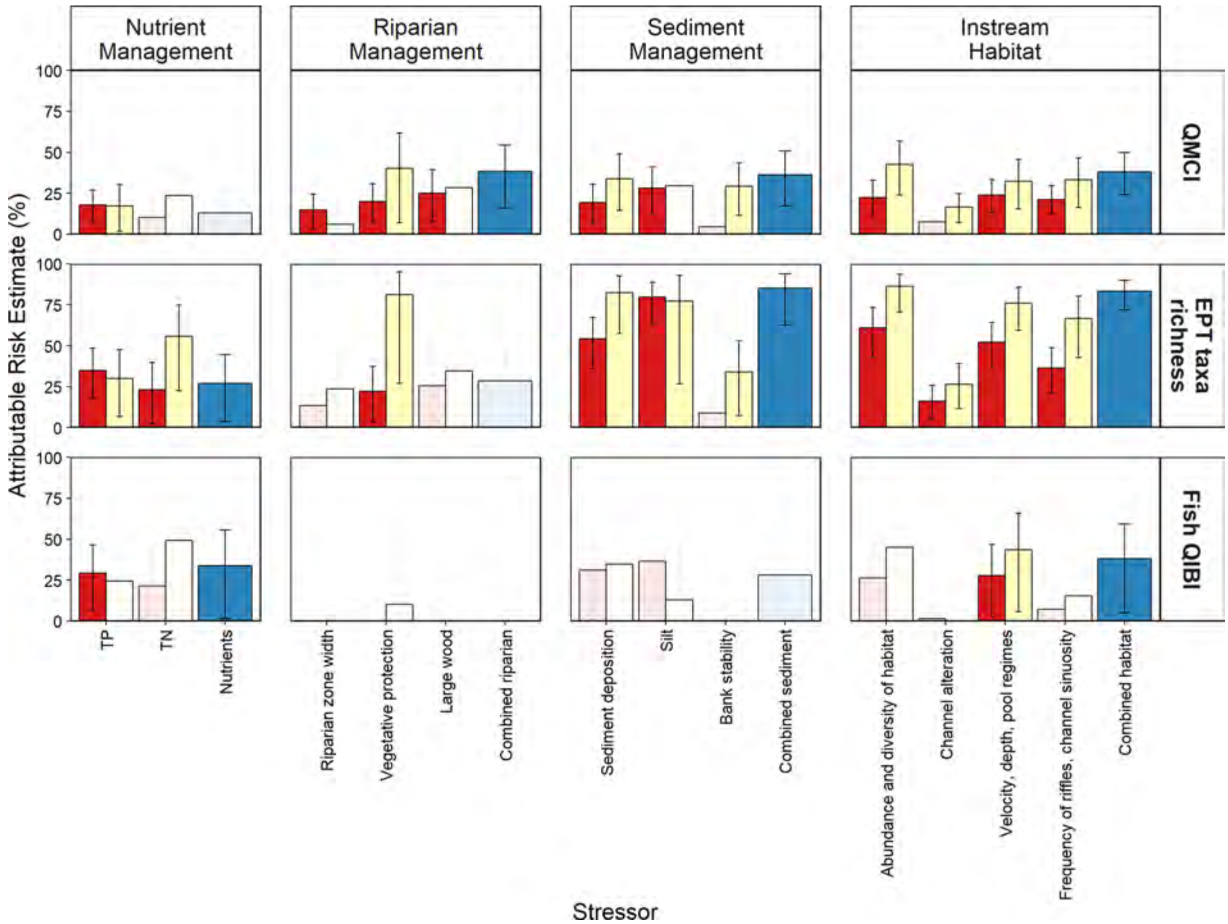

Fig. 5. Mean attributable risk estimates against measured stressors (reduction in Poor biological extent if Poor stressor extent is eliminated) for the Quantitative Macroinvertebrate Community Index (QMCI), EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa richness (excluding Hydroptilidae), and the Fish quantile index of biological integrity (Fish QIBI). Error bars represent 95% confidence intervals. Transparent bars represent attributable risk estimates with a lower 95% confidence interval of < 1%. Red bars represent estimates for Not-poor scenario, yellow bars the Not-Good scenario, and blue bars the combined stressor scenario.

management groups returned attributable risk values with lower 95% confidence intervals of > 1% for the Fish-QIBI (means of 34% and 38%, respectively; equivalent to c.1100–1300 km of wadeable streams; Table 2).

4. Discussion

4.1. Regional stream condition

This study demonstrates the value of a spatially-balanced monitoring network based on probability survey design principles, complemented by an established network of reference sites, to identify management priorities expected to enhance the condition of streams at the regional scale. The environmentally diverse study region in northern New Zealand encompasses c.25,000 km^2, c. 380,000 people and over c.1,250,000 dairy cows (http://www.infometrics.co.nz/ accessed 18 June 2018). Over 50% of the estimated 15,000 km of perennial, non-tidal, wadeable stream length on developed land was subject to some form of biological impairment (Fair or Poor condition). Two-thirds of investigated reach-scale stressors were classed in Poor condition across more than 25% of the network, indicating the likelihood of multiple causal pathways leading to biological impairment of fish and invertebrate communities at a regional scale. Key stressors identified spanned nutrients, sediment, and riparian and instream habitat quality, highlighting a range of interacting factors that could be managed to relieve the Poor biological condition.

4.2. Identification of potentially key stressors

Of the environmental stressors explored, TP, Sediment deposition, % Silt cover, Abundance and diversity of habitat, and Velocity and depth regimes had significant relative risk values for all three biological indices. Invertebrate indices were more likely to be Poor where an individual stressor condition was also assessed as Poor for more than double the number of stressors than the fish QIBI. This potentially suggests that invertebrates provide a more sensitive assessment of reach-scale condition in response to anthropogenic stressors. Possible explanations for this difference include longer life cycles and greater mobility of fish, and unmeasured effects of barriers to longitudinal connectivity throughout the catchment downstream to the sea. New Zealand's native fish fauna is dominated by diadromous species (McDowall, 1990), making them susceptible to anthropogenic barriers disrupting migration (e.g. dams, culverts). Although connectivity was not specifically determined here, it is an important component of the QIBI metric employed (Joy and Henderson, 2007; see Table S1). Furthermore, variation in fish community composition in agricultural catchments elsewhere in New Zealand has been demonstrated to be more affected by landscape scale factors rather than instream variables (Lange et al., 2014). Lange et al. (2014) further hypothesised that the mode of action for nutrient and sediment stressors on fish populations may differ to that for macroinvertebrates.

The co-occurrence of Poor biological and stressor conditions using a range of individually-measured stressor variables is not in itself surprising. Many of the environmental conditions investigated are linked

Table 2
Estimated condition improvements for Quantitative Macroinvertebrate Community Index (QMCI), EPT (Ephemeroptera, Plecoptera and Trichoptera) taxa richness (excluding Hydroptilidae), and the Fish quantile index of biological integrity (Fish QIBI) by management groups, ranked by importance. Values in italics indicate where mean attributable risk estimates had lower 95% confidence interval of < 1%.

Indicator	Extent of Poor biological condition for wadeable streams on developed land	Management group	Extent of Poor stressor condition for wadeable streams on developed land (i.e. amount of network requiring improvement)	Mean attributable risk (% reduction in Poor of biological condition extent if Poor stressor condition improved to at least Fair region-wide)	Estimated reduction in Poor extent of biological condition as length of stream network	Remaining extent of Poor biological condition
QMCI	48.7% c. 7420 km	Riparian management	57.7% (8790 km)	38%	2820 km	4600 km
		Instream habitat management	41.5% (6320 km)	38%	2820 km	4600 km
		Sediment management	55.7% (8500 km)	36%	2670 km	4750 km
		Nutrient management	43% (6560 km)	13%	960 km	6460 km
EPT taxa richness	33.8% c. 5150 km	Sediment management	55.7% (8500 km)	85%	4380 km	770 km
		Instream habitat management	41.5% (6320 km)	83%	4280 km	880 km
		Riparian management	*57.7% (8790km)*	*28%*	*1440 km*	*3710 km*
		Nutrient management	43% (6560 km)	27%	1390 km	3760 km
Fish QIBI	22.5% c. 3430 km	Instream habitat management	41.5% (6320 km)	38%	1300 km	2130 km
		Nutrient management	43% (6560 km)	34%	1170 km	2260 km
		Sediment management	*55.7% (8500km)*	*28%*	*960 km*	*2470 km*
		Riparian management	*57.7% (8790km)*	*0%*	*0 km*	*3430 km*

by current and historical land management practices, for example wholesale land clearance and ongoing conversion to agriculture or urban development which all impact various aspects of stream ecology. The interaction of multiple stressors on biological condition is well documented (Piggott et al., 2012; Townsend et al., 2008). Multiple stressors often interact so that the assessment of risk of a single stressor could be higher or lower than expected (Townsend et al., 2008; Wagenhoff et al., 2012). Irrespective, our results demonstrate that integrated management of multiple stressors is required to improve biological condition at regional scales.

4.3. Priority stressors for policy and intervention

Attributable risk estimates highlight that policy and management interventions aimed at reducing cover by fine sediments, and improving habitat diversity and quality are most likely to reduce the extent of Poor biological condition. The marked association between EPT taxa richness and deposited fine sediment is consistent with experimental studies elsewhere (Elbrecht et al., 2016; Wagenhoff et al., 2012). The condition class threshold, derived from the 99th-percentile at reference sites, used here for silt deposition (c.18%; Table 1) is similar to the 20% fine sediment cover threshold determined by the change-point analysis of Burdon et al. (2013). Wagenhoff et al. (2012), however, identified that to maintain good stream condition based on macroinvertebrates a threshold of 5% fine sediment may be needed. In terms of achieving targets for sediment deposition. Holmes et al. (2016) estimated that, to reduce fine sediment cover of the bed to < 20% for a spring-fed stream in a dairying catchment, an average fenced width of at least 5 m (on both banks of the stream) over at least 300 m upstream would be required.

Attributable risk estimates derived for the combined nutrient management grouping were relatively low (and with lower confidence intervals close to zero) compared to other stressors across all three biological indices (Fig. 5). This finding potentially indicates that, despite their ecological importance, sole management of nutrients cannot be expected to lead to greater ecological improvement at a regional scale than other management approaches. Our water quality data, however, were derived from single samples taken at base flows, and therefore are not necessarily representative of annual nutrient loads. Ecological responses to nutrient reductions in intensive dairying catchments have been shown to be inconsistent (Wright-Stow and Wilcock, 2017). Thus, although often correlated with TN concentrations, changes in invertebrate community composition may be mediated by indirect factors associated with nutrient management, such as periphyton growth (Wright-Stow and Wilcock, 2017).

Common mitigation measures, such as riparian fencing, appear to only partly account for potential improvements in biological indices, indicating the need to integrate fencing with other measures to reach desired policy targets. As indicated in attributable risk analyses, improving riparian vegetation protection to a Good level (covered by mostly native vegetation with reduced disruption) would likely lead to a greater response for macroinvertebrate indices (Fig. 5). Conversely, Poor Fish QIBI scores appear to be insensitive to improvements in riparian condition at a regional-scale, however it is recognised that at a local scale particular native species may respond to riparian condition (Eikaas et al., 2005). Furthermore, the region-wide assessment implemented here is not intended as a substitute for determining specific issues and actions at specific locations (e.g. recruitment issues), as it provides broad scale patterns. Nevertheless, region-wide improvements in riparian vegetation buffers should lead to improvement in the quality of other environmental variables, including reducing sediment inputs and maintaining cooler water temperatures (Piggott et al., 2012), as well as offsetting nutrient effects on periphyton through shading. However, benefits can take a long time to manifest as plants establish and the vegetation canopy matures (Parkyn et al., 2003). Restoration age, however, is not necessarily a good predictor of restoration success, underscoring the importance of catchment scale processes, such as upstream land use, for achieving reach-scale progress towards biological condition targets (Leps et al., 2016). Despite some uncertainty, our results also indicate that improvement to Good for a number of stressors, including TN, Sediment deposition, Bank stability, Abundance and diversity of habitat, and aspects of hydraulic heterogeneity, could further reduce the extent of Poor stream quality based on macroinvertebrate indices. For some relationships, the shift of the stressor threshold value from Fair to Good resulted in less convincing attributable risk estimates, highlighting the need for well-considered alignment of stressor and biological condition classes. Improving the environmental condition of the examined stressors needs to occur at entire catchment or sub-regional scales (e.g. for habitat quality, riparian condition, sediment retention, total nutrients) before local improvements in biological condition can be realised (Death and Collier, 2010; Stoll et al., 2016).

4.4. Stressor management and policy implications

The reversibility assumption of the attributable risk analysis cannot be explicitly validated (Van Sickle and Paulsen, 2008), and recovery trajectories and states often differ from those prior to and during degradation (Harris and Heathwaite, 2012). Nevertheless, such analyses can provide useful guidance to policy writers, decision makers and catchment managers for guiding remediation efforts and setting targets, especially when interpreted in conjunction with relative extent and relative risk estimates (Van Sickle and Paulsen, 2008). For all three biotic indices, the attributable risk estimates of stressors summed to greater than 100%, indicating that there are likely interactions between stressor groups, and that improving the state of one stressor could potentially lead to improvements in other related stressors (Fig. 5). We acknowledge that a limitation of our analyses is that it does not address the interactive effects of stressor conditions on instream ecological values. However, the results highlight that, at a regional scale, multiple stressors will need to be managed to achieve biological improvements in stream condition across the entire network (Table 2). Policy instruments and remediation actions often aim to resolve multiple measures of environmental impairment, or to enhance particular values (Harris and Heathwaite, 2012). Prioritising, integrating and implementing strategies and measures to achieve multiple goals (e.g. improving both fish and macroinvertebrate community condition in our example) are likely to get political support as they are both cost-effective and efficient (Burger, 2008). For example, sediment and nutrients from agricultural run-off can both be partially addressed through the construction or restoration of wetlands (Brix, 1994; Jordan et al., 2003), riparian buffers (Lee et al., 2003), and management of key critical source areas (White et al., 2009). Such measures have potential to contribute to other policy goals, such as enhancing terrestrial biodiversity values (Harris and Heathwaite, 2012). However, additional measures such as restoring fish passage, or control of invasive species, may be required to achieve significant improvements in fish communities (Collier et al., 2017).

A strength of the statistical approach applied here, and of the probabilistic site network employed, is that it provides estimates of uncertainty for extent and risk estimates (e.g. 95% confidence intervals). Communicating and assessing such uncertainty is a key part of the interaction between ecologists, policy makers and decision makers (Rouse and Norton, 2017). Adding to the uncertainty of outcomes, landscape processes, such as the surrounding species pool, regional habitat quality (Sundermann et al., 2011) and the relative positon of streams in the wider dendritic network (Tornwall et al., 2017), will play a role in shaping the biological response to management actions or rehabilitation. Accepting that uncertainty exists (Harris and Heathwaite, 2012) and allowing for indeterminism in decision-making that involves socio-ecological systems (Michaels and Tyre, 2012) are important considerations for policy makers aiming to achieve improved ecological health.

5. Conclusions

While the process used here sets best case scenarios for reversing degradation, it can help to prioritise stressors that need minimum standards set across a wide region by policy makers (i.e. Not-Poor condition class thresholds). Furthermore, the process provides results which can be used to set those standards, and also quantify the spatial extent to which those standards are currently not being achieved. The inter-linked, longitudinal, and multi-level nature of stream and land management can be simplified for policy makers by ranking stressors and encouraging management actions that can lead to improvements in the condition of multiple stressors. One of the strengths of this approach is that, as mechanistic and empirical understandings improve and condition class thresholds for biological responses or stressors are updated, the risk analyses can be repeated. This information should assist decision- and policy-makers to assess the effectiveness of existing approaches, and identify areas that require additional actions or attention (Ministry for the Environment 2018). Looking into the future, ongoing assessment of whether ecosystem health is getting better or worse as a result of investments in resource protection and restoration is also essential (Shapiro et al., 2008).

This study demonstrates that the relative and attributable risk method is an effective way to identify regional-scale stressors on wadeable streams. Based on these findings, management of wadeable streams should be incorporated into policy instruments, such as objectives and rules, which encourage behavioural change amongst resource users, and direct the implementation of environmental mitigations to improve ecosystem health. In particular, this will require the setting of minimum requirements for riparian protection, and nutrient and sediment controls. The diverse geology and climate of the landscape in which this approach was successfully evaluated, supports its general transferability to wadeable streams in contrasting settings. The probabilistic approach utilised here can be applied across a range of natural resources to achieve spatially representative monitoring networks (including freshwater environments such as lakes). The risk approach, based on assessments of biological and stressor conditions allows the identification of priority stressors at regional spatial scales (or greater). The subsequent estimates of potential impact that mitigation actions may achieve, the thresholds involved, and the estimated uncertainty can then be communicated in a manner useful for policy makers. In New Zealand for example, increased uptake of probability-based monitoring networks should aid in improving the overall targeting of policy and implementation to be more biologically meaningful. Such policies will likely require integrated management of multiple stressors. The case study detailed here shows improving management of sediment, riparian zones and instream habitat is most likely to enhance overall biological condition across the region, highlighting the need to extend policy development and implementation beyond a singular focus on water quality if ecosystem health objectives are to be met.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.envsci.2018.11.028.

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