The Impact of Deer Farming on the Ecological Health of New Zealand Streams

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Charlotte Claire Minnis

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Abstract

There is limited research on what water chemistry and habitat factors of streams may be affected by deer farming in New Zealand and thus impact their ecological health. Nine high country and hill country deer farms across New Zealand were assessed using freshwater macroinvertebrate samples, nutrient data, periphyton biomass and visual habitat assessments. Biological metrics calculated from the macroinvertebrate samples indicated that the waterways on these deer farms had moderate to severe impairment. This impairment, as well as the community composition, was linked most strongly with the levels of deposited sediment found at the sites. Deposited sediment was clearly the most significant impact on the ecological health of streams of these high country and hill country deer farms. The Stream Health Check is a tool developed to help landowners assess the quality of their waterways while also providing education and environmental awareness. The purpose of this study was to investigate whether the score from a Stream Health Check assessment correlated with freshwater macroinvertebrate biological metrics of the site. The scores from the Stream Health Check correlated with the Macroinvertebrate Community Index (MCI) and the Quantitative Macroinvertebrate Community Index (QMCI) of deer farm streams and also streams in mixed agricultural practice catchments. This suggests that the Stream Health Check would be a useful tool for landowners in the development of Environmental Farm Plans as it is an inexpensive way to assess the physical and ecological quality of a waterway that relates well to more scientific methods for assessing ecological health.

Introduction

Agricultural has the potential to have a detrimental impact on the chemical, physical and ecological health of waterways. The first chapter of this thesis reviews the current literature on the impacts of different forms of agriculture on water quality and ecological health. It also reviews some of the mitigation practices used in New Zealand to reduce waterway impairment caused by agricultural land use.

The second chapter of this thesis investigates which chemical and physical factors are driving the ecological health of waterways draining deer farms. There are currently limited studies on the impacts of deer farming, in particular the impact on freshwater macroinvertebrate communities. In this study I sampled the freshwater macroinvertebrates on nine deer farms across New Zealand. The biological metrics and community composition were then compared to nutrient levels, periphyton biomass and deposited sediment cover.

The third chapter of this thesis investigates whether the Stream Health Check score correlates to biological metrics of freshwater macroinvertebrates. The Stream Health Check is a tool designed to help landowners assess the quality of their waterways and identify areas that should be prioritised for mitigation. The collection and processing of macroinvertebrates as well as the description of the nine deer farms throughout New Zealand were the same as in Chapter 2 and so the methods and site descriptions have been repeated in Chapter 3.

Chapter One

Literature review of the effects of agriculture on New Zealand benthic macroinvertebrate communities

Introduction

Management of freshwater in areas with agricultural land use has become an issue of concern amongst both the farming fraternity and general public. Agriculture is one of the most pervasive anthropogenic activities affecting running waterways (Vörösmarty et al., 2010). In fact, it has been suggested that intensive agriculture is one of the greatest threats to the health of running waterways globally. Intensive agriculture induces ecosystem degradation, correlative to the alteration of the physical habitat and water chemistry of a waterway (Harding & Winterbourn, 1995; Malmqvist & Rundle, 2002; Quinn, 2000). Farming animals at high stock densities can result in increased organics, periphyton and sediment, and reduced riparian vegetation and flow in rivers and streams (Larned et al., 2004; Malmqvist & Rundle, 2002; Quinn, 2000; Quinn & Hickey, 1990a). These environmental variables can impact the ecological health of a waterway and macroinvertebrate communities (Hall et al., 2001; Quinn & Hickey, 1990a).

Physical habitat and water chemistry are strongly influenced by land use and can interact as environmental stressors (Townsend et al., 2008; Wagenhoff et al., 2011). Riparian vegetation has been found to improve bank stabilisation, provide shade, intercept nutrient run-off and reduce microbial contamination (Collier, 1995; Collins et al., 2007; Dosskey et al., 2010; Greenwood et al., 2012; Parkyn, 2004; Wilcock et al., 2009). This results in streams with riparian vegetation having lower water temperatures, less deposited sediment and reduced periphyton growth (Greenwood et al., 2012). Phosphorus and nitrogen are often elevated in streams with pastural land use due to fertiliser loss and livestock inputs (Quinn, 2000). These elevated nutrient levels can also influence excessive periphyton growth (Liess et al., 2012). While periphyton is naturally present in streams and is a resource for macroinvertebrates, excessive blooms can be detrimental to the health of the waterway (Liess et al., 2012; Marsh, 2012; Tilman, 1999). Periphyton can increase fluctuations in dissolved oxygen and pH and cause interstitial anoxia in the stream bed (Quinn & Gilliland, 1989; Quinn & Hickey, 1990b) . Fine deposited sediment can also cause interstitial anoxia and habitat loss as it fills the interstices of gravel beds (Dolédec et al., 2006; Ryan, 1991). Deposited sediment impacts substrate availability for periphyton growth and can therefore restrict the organic resources available to macroinvertebrates (Broekhuizen et al., 2001; Liess et al., 2012). This can result in a reduction of grazers such as *Deleatidium* (mayfly) and other EPT taxa and has been found to reduce overall taxon richness (Broekhuizen et al., 2001; Wagenhoff et al., 2011).

Changes in these environmental variables as the result of agriculture can cause a shift in macroinvertebrate community structure (Hall et al., 2001; Quinn & Hickey, 1990a). Macroinvertebrate communities in agriculturally polluted systems have reduced numbers of sensitive EPT taxa (Ephemeroptera, Plecoptera and Trichoptera) and an increase in pollution tolerant taxa such as Chironomidae, Mollusc, and Oligochaeta (Hall et al., 2001; Harding & Winterbourn, 1995; Quinn, 2000; Quinn & Hickey, 1990a; Scott et al., 1994). The intensity of the agricultural activity in a catchment can also determine the magnitude of these changes in the community (Davies-Colley & Nagels, 2002; Hamill & McBride, 2003; Quinn, 2000). Catchments with greater than 30% of their area in pasture have been observed to have reduced macroinvertebrate community diversity; and up to a 40% reduction in their Quantitative Macroinvertebrate Community Index (QMCI) score and a 6-fold reduction in the number of EPT taxa (Quinn, 2000). The geology of a catchment can also influence the magnitude of potential effects caused by agricultural activities. Agriculture in a catchment with erosion prone geology will see greater sedimentation and habitat degradation then agricultural land use in catchments with harder geology (Young et al., 2005). Several studies have found that land use effects vary with geology (Quinn & Hickey, 1990a; Young et al., 2005). Thus, not all catchments and their respective invertebrate communities respond to agriculture the same way.

Dairy Farming

Dairy farming has the biggest stock per unit impact on freshwater quality in New Zealand (Foote et al., 2015; Joy, 2015; Quinn et al., 2009). Numerous studies demonstrate that intensive dairy farming causes eutrophication and degradation of rivers and streams (Davies-Colley & Nagels, 2002; Foy & Kirk, 1995; Hamill & McBride, 2003; Hooda et al., 2000; Quinn et al., 2009; Scarsbrook & Melland, 2015; Schofield et al., 1990; Wilcock et al., 2007). Dairy farming can cause large increases in nitrogen, phosphorus, sediment and faecal bacteria such as E. coli as well as decreased levels of dissolved oxygen (Chalar et al., 2017; Hooda et al., 2000; Scarsbrook & Melland, 2015; Wilcock et al., 2007). The sources of these inputs are surface run-off, soil leeching (Monaghan et al., 2016) or stock with direct access to the waterway (Chalar et al., 2017; Kyriakeas & Watzin, 2006). Waterways that have stock access tend to have embedded substrate, higher deposited sediment and nutrient levels then waterways where stock are excluded (Kyriakeas & Watzin, 2006). Without stock exclusion grazing occurs in the riparian margin, reducing riparian vegetation which in turn increases bank erosion and reduces instream habitat and shading of the waterway (Greenwood et al., 2012; Quinn et al., 1992).

The impacts of dairy farming have often been found to be greater than other livestock farming such as sheep and beef (Quinn et al., 2009; Quinn et al., 1992; Wagenhoff et al., 2012). Dairy farms contribute to high levels of nutrient input proportional to the percentage of the catchment area being dairy farmed (Elliott et al., 2005). Livestock urine can account for up to 90% of the nitrogen leaching (De Klein, 2001; Ledgard et al., 2009), but there are factors other than the direct impact of dairy cows themselves. Chalar et al. (2017), found that increased nutrient levels were correlated with intensive fertiliser regimes used in dairy farming to maintain the forage crops and pasture growth necessary for the intensive grazing systems. Nutrient and sediment exports were also correlated with inadequate treatment of shed effluent and high stocking rates (Chalar et al., 2017). Catchments without intensive dairy farming had better water quality, with lower exports of nutrients and fine sediment (Aarons & Gourley, 2013; Chalar et al., 2017).

The issues caused by dairy farming can be exacerbated further when cows have direct access to waterways (Kyriakeas & Watzin, 2006; Quinn et al., 1992). This contributes to increased deposited sediment, substrate embeddedness and direct nutrient input from faecal and urine contamination (Kyriakeas & Watzin, 2006). The effect of this on the riverbed causes a loss of available habitat between and beneath the substrate (Burdon et al., 2013). Urine and faeces act a point source for nutrients and faecal bacteria. Davies-Colley et al. (2004), found that when crossing a ford, cattle had 11 minutes access to the waterway which resulted in increased turbidity, nitrogen and phosphorus loading, and *E.coli* contamination was 100 times higher.

Given these impacts it is not surprising macroinvertebrate communities are also affected (A. E. Wright-Stow & R. J. Wilcock, 2017). Waterways with high levels of nutrients and deposited sediment have been found to have low EPT taxon richness and total taxon richness (Matthaei et al., 2006; Townsend et al., 2008). Ramezani et al. (2016), found that

EPT taxa richness, total taxon richness and the Macroinvertebrate Community Index (MCI) decreased along a dairy farming gradient (from 0% to 80% catchment in dairy). Pollutant sensitive taxa *Deletidium* (single-gilled mayfly)*, Pycnocentrodes* (cased caddis) and *Pycnocentria (*cased caddis) all declined with increased dairy farming (Ramezani et al., 2016). Pollutant tolerant taxa *Paracalliope* (crustacean*), P. acuta, P.antipodarum* (Mollusca) and *Austrosimulium spp.* (Diptera) responded positively to increased dairying (Ramezani et al., 2016). Oligochaeta increased in catchments with between 40% and 80% dairying (Ramezani et al., 2016). Dairy farm streams with no active mitigation practices have degraded water quality which is reflected in low MCI scores and macroinvertebrate diversity (A. E. Wright-Stow & R. J. Wilcock, 2017). A study of Welsh dairy farms found that the impacts on invertebrate communities were not always consistent across the whole catchment. They found that the most severely impacted waterways were ones close to high intensity grazing areas and next to raceways, reflected by low diversity communities dominated by pollutant tolerant *Chironomidae sp.* (Schofield et al., 1990).

Sheep and Beef Farming

The effect of sheep and beef farming on the ecological health of streams is often assessed by comparing them to native forest streams (Collier et al., 2000; Hall et al., 2001; Harding & Winterbourn, 1995; Scott et al., 1994). Sheep and beef farming has been found to cause nutrient enrichment, increased sediment and temperature, with decreases in riparian vegetation and altered macroinvertebrate diversity (Hall et al., 2001; Harding & Winterbourn, 1995; Quinn & Hickey, 1990b). However, when compared to dairy and deer farming, streams on sheep only farms have lower levels of sedimentation, nitrogen and phosphorus (Matthaei et al., 2006; McDowell & Wilcock, 2008). Sheep and beef farming has greater impacts than sheep only farming, but overall has much lower impacts than deer and dairying (McDowell & Wilcock, 2008). Scott et al. (1994), found sheep only grazing did not result in elevated phosphate and nitrogen levels but did cause an increase in the deposition of fine sediment. They also found that higher intensity sheep grazing did not cause an increase in nutrient levels (Scott et al., 1994).

Although sheep and sheep and beef farming, have less impact on water quality than dairy farming, macroinvertebrate composition is still impacted (McDowell & Wilcock, 2008; Quinn et al., 2009). Communities monitored in pastural streams grazed by sheep and/or sheep and beef have lower MCI and QMCI scores and proportions of EPT taxa than streams in the same catchment draining native vegetation (Collier et al., 2000; Hall et al., 2001; Harding & Winterbourn, 1995; Quinn & Hickey, 1990a; Scott et al., 1994). Macroinvertebrate communities in streams on sheep and beef farms are often dominated by pollution tolerant species such as *P. antipodarum* (Hall et al., 2001; Wagenhoff et al., 2012), with sheep and beef streams having significantly fewer EPT taxon then streams in native forest (Collier et al., 2000; Quinn & Hickey, 1990a). Scott et al. (1994) found that in areas grazed by sheep, the QMCI scores were low, with community composition dominated by pollutant tolerant taxa groups such as Mollusca and Oligochaeta. In areas where grazing was absent QMCI scores and EPT taxon richness were higher (Scott et al., 1994). Contrastingly, Riley et al. (2003), found that the MCI and QMCI of naturally nutrient depleted streams in Otago increased with low intensity sheep and beef farming when compared to native bush in the same catchment. It was suggested that increased nutrient inputs caused an increase in primary productivity resulting in increased macroinvertebrate biodiversity (Riley et al., 2003).

Deer Farming

The impacts of deer farming on freshwater systems are largely unexplored (De Klein et al., 2003; Quinn, 2000), however it is postulated high intensity deer farming, like dairy farming, would have a greater impact than sheep farming (McDowell & Wilcock, 2008; Quinn et al., 2009). Several studies have found the level of phosphorus enrichment and sedimentation caused by deer farming is greater than that of dairy farming (Matthaei et al., 2006; McDowell & Wilcock, 2008). Nitrogen loss in contrast was lower than from that of dairy farms but higher than from sheep and beef farms (McDowell & Wilcock, 2008).

Increased deposited sediment appears to be the most prevalent environmental impact caused by deer farming (McDowell, 2008, 2009a, 2009b; Pattis et al., 2017; Rhodes et al., 2007; Sakai et al., 2012). It is assumed that the wallowing activity of deer would increase sediment deposition in waterways (Quinn, 2000). Deer create wallows in hollows in the ground near waterways or in shallow waterways themselves (McDowell, 2009b; Pattis et al., 2017). These wallows are one of the main sources of increased sediment, nutrients, and faecal run-off (De Klein et al., 2003; McDowell, 2009a, 2009b; Pattis et al., 2017). Increased sediment, nutrients and faecal contamination have also been linked with the fence-pacing behaviour of deer (De Klein et al., 2003; McDowell, 2008; McDowell & Paton, 2004; Pattis et al., 2017). Fence-pacing is observed frequently in farmed deer and can cause loss of pasture and soil compaction, this in turn causes increased run-off and soil erosion, resulting in degradation of surrounding waterways (McDowell, 2008; Moore et al., 1985). When deer are excluded from waterways sediment levels decline with time (McDowell, 2009b; Sakai et al., 2013; Sakai et al., 2012). Increased sediment in streams results in a loss of habitat from the filling of interstitial spaces in the bed substrate, smothering periphyton, mosses and macroinvertebrates (Broekhuizen et al., 2001; Dolédec et al., 2006). Decreased stream moss growth as a result of sediment deposition caused by deer has been observed in streams in both Japan and Otago, New Zealand (Matthaei et al., 2006; Sakai et al., 2012).

There are limited studies on how farmed deer impact aquatic macroinvertebrate communities (De Klein et al., 2003). In a study of 30 Southland deer pasture streams it was found many of the streams had high fine sediment and low bank cover (Rhodes et al., 2007). All streams had a QMCI less than 5 indicating moderate pollution, with 13 of the streams with scores less than 4 indicating severe pollution (Rhodes et al., 2007). Sakai et al. (2013), found that when wild Sika deer in Japan were excluded from streams, taxonomic richness of macroinvertebrates in first and second order streams increased (Sakai et al., 2013). This increase was correlated with decreased sedimentation and increased periphyton biomass (Sakai et al., 2013). The number of Ephemeroptera and Trichoptera species was also higher in catchments with deer exclusion, while the proportion of Gastropods was higher in catchments where deer still had stream access (Sakai et al., 2012). The taxonomic limitation seen in catchments with deer was thought to be caused by the impacts of excessive fine sediment deposition (Sakai et al., 2013). In regions with deer there was evidence of bank trampling and minimal understory vegetation causing bank erosion (Sakai et al., 2013; Sakai et al., 2012). These catchments had double the fine sediment cover compared to the neighbouring catchments with deer exclusion (Sakai et al., 2013; Sakai et al., 2012). The increased sediment was correlated with the three fold reduction in periphyton growth compared to the exclusion catchment (Sakai et al., 2013; Sakai et al., 2012). Although these two studies looked at wild deer, they illustrate the impact deer can have when they have access to aquatic systems.

Mitigation

The mitigation of agricultural effects on water quality and ecological health has been an area of interest that has continued to grow as new regulations for water quality indicators have been introduced. There are a range of commonly used mitigation practices such as restricting stock access to streams with fencing, riparian afforestation, erosion control planting and improved effluent, irrigation, and fertiliser management (Monaghan et al., 2021). The aim of these mitigation practices is to reduce the nutrient and sediment inputs into waterways with the intention of improving the water quality and overall ecological health. These mitigation practices are not always effective, and the ecological recovery is often delayed in comparison to improvements in water quality (Hamilton, 2012; Parkyn, 2004). Ecological recovery is limited by the connectivity to surrounding unimpacted communities for recolonisation once the water and habitat quality has improved (Greenwood et al., 2012; Harding et al., 1998). The fragmented use of mitigation practices throughout a catchment may not be overly successful in improving ecological health and it has been suggested that catchment wide restoration may be necessary (Greenwood et al., 2012; Harding et al., 1998). Both nutrient and sediment inputs need to be controlled to mitigate stream health degradation, though it has been suggested that deposited sediment may be the more impactful variable and should perhaps be prioritised (Wagenhoff et al., 2012).

Fencing waterways to remove stock access is a mitigation practice that is used to improve the quality of the riparian margin and decrease the sediment loading caused by erosion and bank trampling (Greenwood et al., 2012; McDowell, 2008; Parkyn, 2004; Quinn, 2000). It also decreases the direct input of nutrients into a waterway from faecal matter and urination but has a limited impact the pastoral run-off. (Aarons & Gourley, 2013; Quinn, 2000). For this reason, it has been argued that with high intensity practices, such as dairy farming, that fencing exclusion is not sufficient because of the excessive volume of surface run-off and soil leeching (A. E. Wright-Stow & R. J. Wilcock, 2017). Stock exclusion used in conjunction with riparian revegetation though has been found to reduce nutrient run-off as well as microbial contamination (Collins et al., 2007; Parkyn, 2004). Riparian vegetation also reduces stream bank erosion, increases shading and improves the habitat available for both terrestrial and aquatic organisms (Greenwood et al., 2012; Wilcock et al., 2009).

The effective management of the timing, volume and location of effluent irrigation can reduce the level of nutrient run-off and microbial contamination (Collins et al., 2007). By only irrigating when the soil water capacity will not be exceeded reduces the loss of nutrients to waterways (Houlbrooke et al., 2004) The density at which livestock are farmed also has an effect on the level of environmental consequences (Davies-Colley & Nagels, 2002; Riley et al., 2003). In particular the increased stocking rate of dairy cows has been linked to declined water quality (Hamill & McBride, 2003). The reduction of stocking rates, especially during times of high rainfall, would be an effective way to reduce the environmental impacts of agriculture (Cournane et al., 2011).

Conclusion

Agricultural practices influence the nutrient concentrations of the water, the health of the riparian margin, levels of deposited sediment and periphyton biomass in adjoining waterways (Foy & Kirk, 1995; Harding et al., 1999; Quinn, 2000). These variables interact to influence habitat quality available for instream life (Townsend et al., 2008). As a result, macroinvertebrate assemblages are often detrimentally impacted by intensive animal agricultural, and this is often reflected in the reduction of taxonomic richness and pollution sensitive taxa and low MCI or QMCI scores (Hall et al., 2001; Hamill & McBride, 2003; Quinn & Hickey, 1990a). The form of agricultural and mitigation practices in place can influence the degree to which these communities are impacted (Davies-Colley & Nagels, 2002; Quinn, 2000). The level of effect of dairy farming and sheep and beef farming on freshwater is thoroughly researched, but deer farming has received considerably less attention (De Klein et al., 2003). Based on available limited studies on deer farming it is suggested that increased deposited sediment is the largest impact of farmed deer on freshwater (McDowell & Wilcock, 2008; Quinn, 2000; Rhodes et al., 2007). As a result of high levels of sedimentation sediment tolerant taxa such as Potamyprygus are expected to dominate macroinvertebrate communities, with MCI and QMCI scores indicating moderate to severe pollution (Rhodes et al., 2007; Sakai et al., 2012).

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Chapter 2

What determines ecological health in streams on deer farms?

Abstract

There is limited research on what water chemistry and habitat factors of streams may be affected by deer farming in New Zealand and thus impact their ecological health. Nine high country and hill country deer farms across New Zealand were assessed using freshwater macroinvertebrate samples, nutrient data, periphyton biomass and visual habitat assessments. Biological metrics calculated from the macroinvertebrate samples indicated that the waterways on these deer farms had moderate to severe impairment. This impairment was linked most strongly with the levels of deposited sediment found at the sites. The community composition of the freshwater macroinvertebrates found across the nine sites was also influenced predominantly by the level of deposited sediment. Nutrient concentrations in the water appeared to have no relationship with the biological metrics but did have an impact on the community composition across the sites. Deposited sediment was clearly the most significant impact on the ecological health of streams of these high country and hill country deer farms.

Introduction

Waterways throughout the globe are being adversely affected by increases in agricultural intensification (Vörösmarty et al., 2010). There is a growing urgency for management options to ameliorate some of these adverse environmental impacts from agriculture. The impacts of dairy, sheep and beef farming on freshwater systems have been widely investigated but the impacts of farming deer have not been investigated as thoroughly. In 2020 there were 1,400 deer farms in New Zealand, running a total of 833,000 deer (Deer Industry New Zealand, 2020). The deer industry needs a better understanding of the ecological health of waterways draining deer farms and the potential mitigation options available to ameliorate any adverse effects.

To date the limited research on deer farm waterways suggests that deer cause an increase in fine sediment deposition from wallowing and fence pacing behaviours (De Klein et al., 2003; McDowell, 2008; McDowell & Paton, 2004; Pattis et al., 2017; Quinn, 2000). Deer form wallows in land hollows near waterways or lie directly in the flowing water. This natural wallowing behaviour is a source of sediment, nutrients, and faecal contamination (De Klein et al., 2003; McDowell & Paton, 2004; Pattis et al., 2017). Deer with access to waterways and the riparian margin also cause riparian damage, soil compaction and increased bank erosion through grazing and bank trampling (McDowell, 2007, 2008; McDowell & Wilcock, 2008; Sakai et al., 2012). Deer farming has been found to have lower nitrogen loss than that of dairy farming but higher levels of deposited sediment and dissolved phosphorus (McDowell & Wilcock, 2008; Quinn et al., 2009). High levels of deposited sediment is well known to have numerous detrimental impacts on in-stream biological communities (Burdon et al., 2013; Quinn et al., 2009; Quinn, 2000). Rhodes et al. (2007) found that across 30 deer farms in New Zealand, all had a Quantitative MCI (QMCI) score less than 5.0, indicating moderate ecological impairment. Instream periphyton and moss growth is lower, and taxonomic richness of macroinvertebrates is reduced because of the high levels of deposited sediment in waterways that deer can access (Rhodes et al., 2007; Sakai et al., 2013; Sakai et al., 2012).

In this study, I investigated the environmental variables influencing the ecological health of waterways directly impacted by high-country and hill-country deer farming. I hypothesise that the ecological health will be higher in streams with lower deposited sediment cover and streams with higher levels of deposited sediment will have freshwater invertebrate communities dominated by tolerant taxa such as Molluscs and Oligochaeta.

Study area

High country and hill country deer farming is found throughout New Zealand. The 9 sampled sites are deer farms in the Hawkes Bay, Waikato, Marlborough, Tasman, Otago and Southland regions (Figure 2.1).

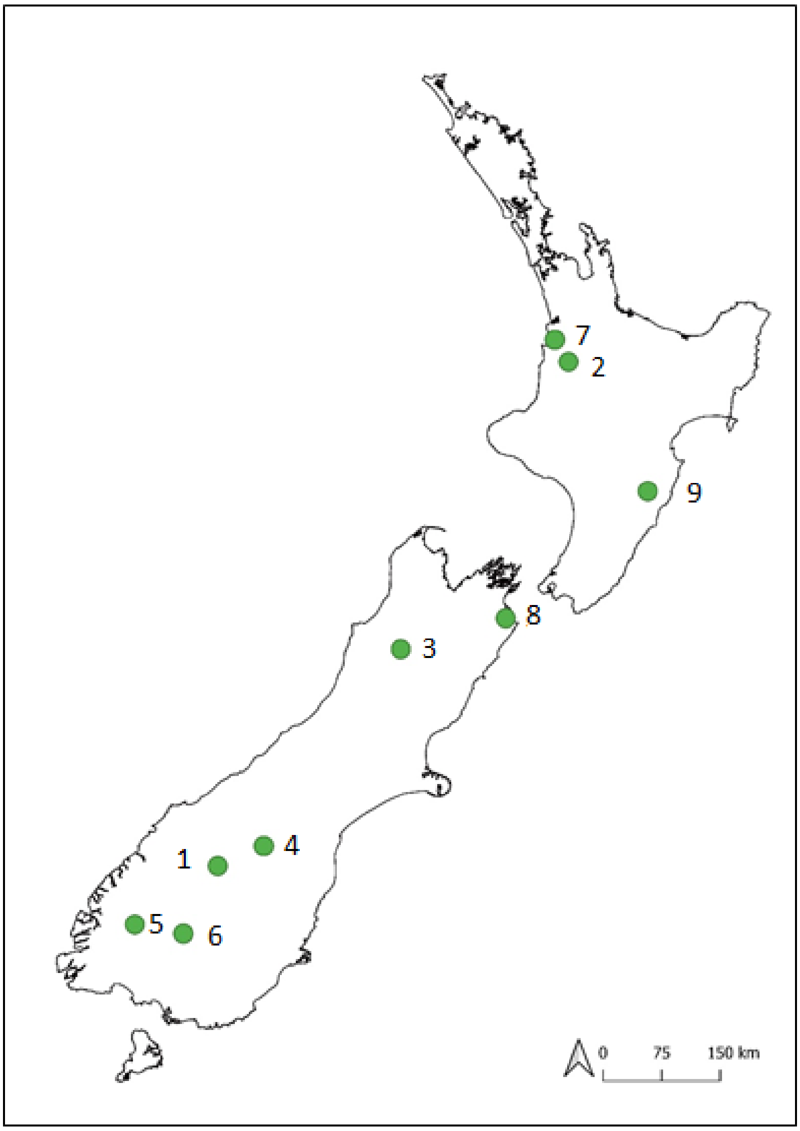
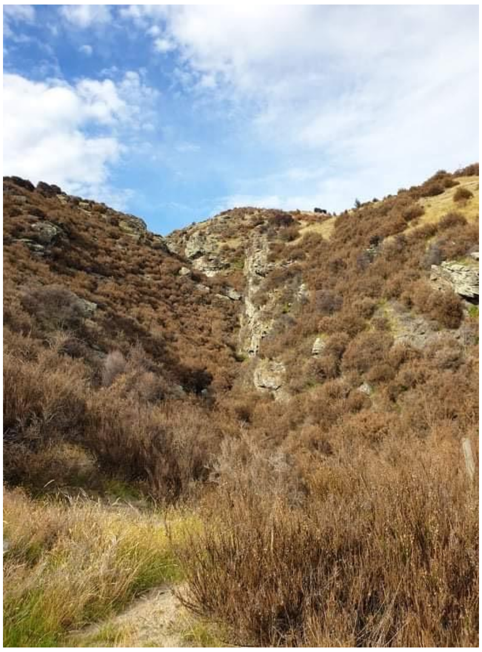


Figure 2.1. Locations of the nine high and hill-country deer farms around New Zealand

Site 1: A 2000-hectare block of tussock dominant high country in Otago. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above between November and February of each fawning season. The site had historically been used for gold mining.



Site 2: A 770-hectare block of rolling hill country in Waikato. The sampling site had no stock exclusion fencing with beef cattle and deer in the catchment above the site. The site had evidence of bank tramping. The stream was a stony bottomed.

Site 3: A 440-hectare block of river flats and terraces in Tasman. The sampling site had no stock exclusion fencing with deer breeding stock in the catchment above.

Site 4: A 3500-hectare block of medium high country in Otago. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above. The site had some bank erosion present due to slips.

Site 5: A 5500-hectare block of tussock dominant hill country in Southland. The sampling site had no stock exclusion fencing and the catchment above is used for fawning.

Site 6: A 1570-hectare block of steep to medium hill country in Southland. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above. The site had schools of Gollum Galaxiid present.

Site 7: A 980-hectare block of steep hill country in Waikato. The sampling site had no stock exclusion fencing and had a mix of livestock classes. This catchment represents the highest rainfall catchment of the 9 sites. Koura were also present during invertebrate sampling.

Site 8: A 180-hectare block of flat to rolling hill country in Marlborough. The sampling site had stags wintering in the paddock with no stock exclusion fencing. The site had substantial bank erosion and slow flowing water with reaches of the stream drying up fully in summer.

Site 9: A 550-hectare block of steep hills to rolling country in Hawkes Bay. The sampling site had no stock exclusion fencing with only deer in the catchment above the site. The site had evidence of bank tramping and high levels of deposited sediment.

Materials and methods

Macroinvertebrate assessments

Samples were collected at each site between February and March, 2021. Macroinvertebrates were collected from riffles and runs within a 100 m reach of the stream using a 0.5 mm mesh Surber sampler. At each site five Surber samples were collected and preserved in 90% ethanol. The samples were processed in the laboratory using a 200 individual fixed count and scan for rare taxa (Stark et al., 2001). Invertebrates were identified to genus level using Winterbourn et al. (1989).

A range of biological indices were calculated from each sample, including the MCI (Macroinvertebrate Community Index), the QMCI (quantitative MCI) and ASPM (Macroinvertebrate Average Score Per Metric) (Clapcott et al., 2017; Collier, 2008; Stark & Maxted, 2007). The taxa tolerance scores described in Stark and Maxted (2007); Winterbourn et al. (1989) were used for calculating the MCI and QMCI scores. When normalising the data for the ASPM calculations, the national maximum and minimum were applied (Environment, 2020).

Habitat and Periphyton sampling

Periphyton samples were collected at sites when appropriate substrate was present (Sites 8 and 9 had no suitable substrate). Chlorophyll *a* was extracted from 5 stones at each site using 90% acetone at 5°C. After 24hr in the dark absorbances were read on a Cary 50 Conc UV-Visible spectrophotometer™ and converted to pigment concentrations following Steinman et al. (2017). The surface area of the stones was estimated using axial measurements following Graham et al. (1988)**.** The calculated surface areas were then halved as only the top half of the stone is available for periphyton growth. A visual estimate of the percent of deposited sediment coverage was made following Clapcott et al. (2011), as well as a visual assessment of the proportion of each substrate category present based of the sizing outlined in Wolman (1954). A stream health check was completed at each site (Polglase, 2000). The stream health check scores are based off four categories: the quality of the riparian margin, instream life, potential for contaminants, and the stream channel quality.

Water chemistry

Water samples for nutrient analysis were not collected at the time of macroinvertebrate sampling. However, the sites have had water samples collected monthly since 2017. Medians of nutrients in the water samples collected in 2021 were used in this study.

Data Analysis

Linear regression models were fitted to investigate relationships between the biological metrics, nutrients, sediment and periphyton. Non-metric multidimensional scaling (NMDS) using the Hellinger distance measure was used to investigate patterns in community structure between the nine sites. All statistical analysis was completed using Rx64 4.0.2 (R Core Team, 2022).

Results

Biological indices

Based on their biological metrics the sampled sites had moderate to poor ecological health (Table 2.1). Four sites had MCI scores below the National Policy Statement Freshwater Management (NPSFM) national bottom line, indicating severe impairment (Ministry for the Environment, 2020). The other five sites had scores that ranged from mild to moderate impairment (Ministry for the Environment, 2020). Five sites had QMCI scores below the national bottom line but only two had ASPM scores sat below the national threshold (Ministry for the Environment, 2020).

Table 2.1. The averages for the biological metrics, sediment cover (%), Periphyton biomass (mg chl-a/m^2), Total Nitrogen (TN), Nitrite Nitrate Nitrogen (NNN) and Dissolved Reactive Phosphorus (DRP)

	MCI	QMCI	ASPM	Sediment (% cover)	Periphyton (mg chl-a/m^2)	TN (mg/L)	NNN (mg/L)	DRP (mg/L)
Site 1	112.7	6.4	0.70	5	219.0	0.89	0.51	0.02
Site 2	101.2	6.4	0.59	50	299.8	2.04	1.85	0.01
Site 3	108.7	3.7	0.69	10	171.4	0.09	0.01	0.02
Site 4	84.3	2.4	0.38	80	169.6	0.12	0.01	0.01
Site 5	97.1	4.5	0.50	55	24.5	1.11	0.57	0.01
Site 6	93.1	4.5	0.51	70	60.3	0.25	0.05	0.01
Site 7	81.8	3.5	0.40	15	142.4	0.88	0.77	0.01
Site 8	71.2	4.1	0.25	100	n/a	1.31	0.86	0.02
Site 9	67.1	2.8	0.22	100	n/a	0.81	0.02	0.01

There were no significant relationships between the three biological indices and any of the nutrient measures or periphyton biomass (Appendix 1). On the nine deer farms sampled, 6 of these had levels of deposited sediment cover above 50% (Fig 2.1). All three biological metrics declined with increasing deposited sediment cover ($F_{(1,43)} = 45.06$, $P<0.001$, $r^2= 0.5$, $F_{(1,43)} = 7.584$, $P<0.05$, $r^2 = 0.13$ $F_{(1,43)} = 40.8$, $P<0.001$ $r^2 = 0.47$, MCI, QMCI and APSM respectively) (Fig. 2.2).

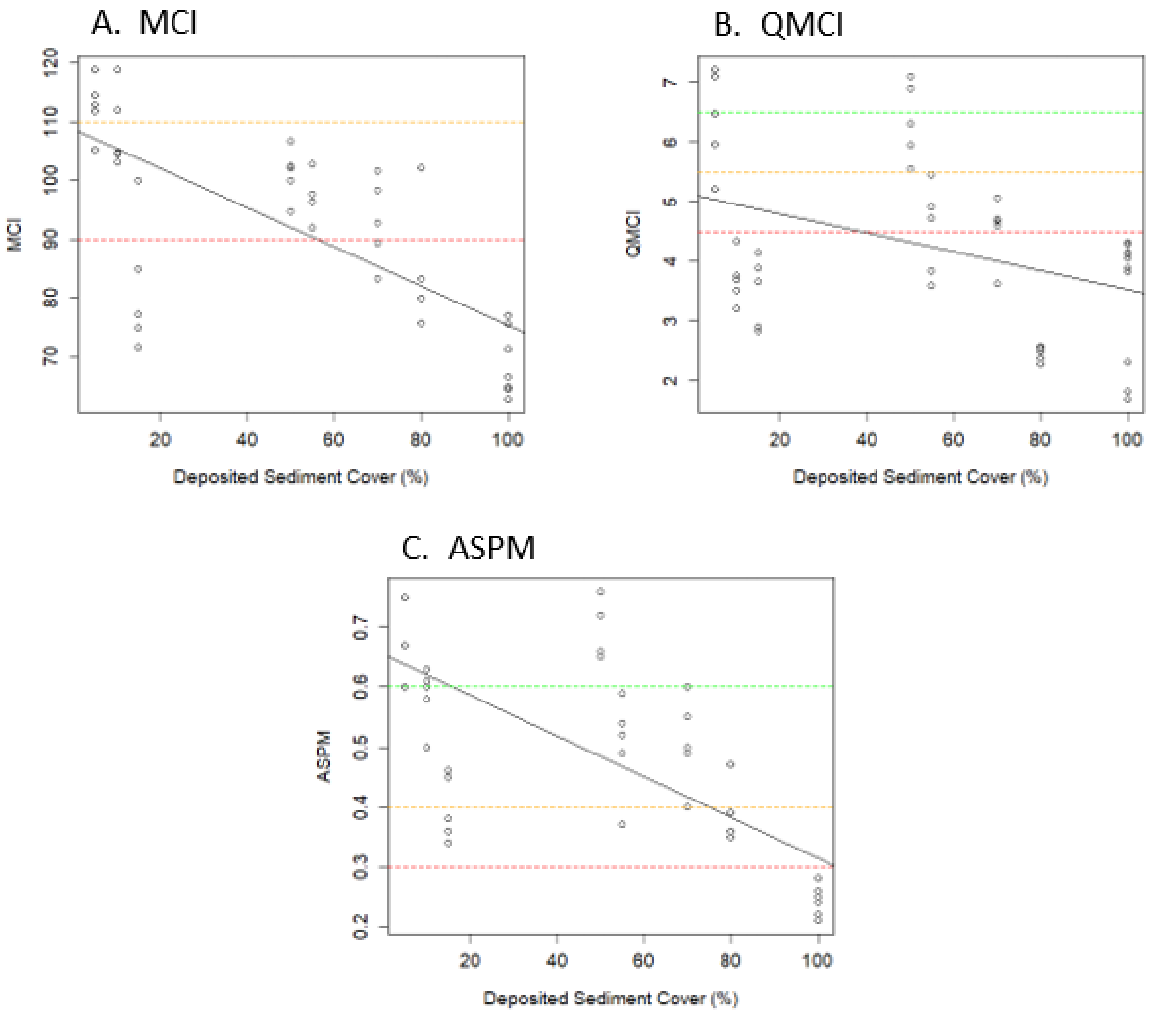


Figure 2.2. Plot of A. MCI, B. QMCI and C. ASPM as a function of deposited sediment cover collected at 9 deer farm streams in January 2021. Data points are Surber samples. The red dotted line indicates the national bottom line set in the NSPFM, the yellow dotted line indicates the lower limit for class B and the green dotted line indicates the lower limit for class A.

Community Composition

Invertebrate communities at sites with high levels of deposited sediment were dominated by the mollusc *Potamopygrus antipodarum* and Oligocheta (Fig 2.4). In contrast sites with low levels of deposited had more diverse invertebrate communities dominated by *Deleatidium* and *P. antipodarum.*

Ordination of the invertebrate communities graded the site communities along axis one from sites with low levels of sediment to ones with high levels (Fig. 2.3). Although, these were also sites with high and low biomass of periphyton, respectively, and clearly both environmental drivers are linked (Table 2.1). The stress of 0.05 indicated a good ordination. Axis two was positively correlated with nitrogen concentrations and negatively with stream size (Table

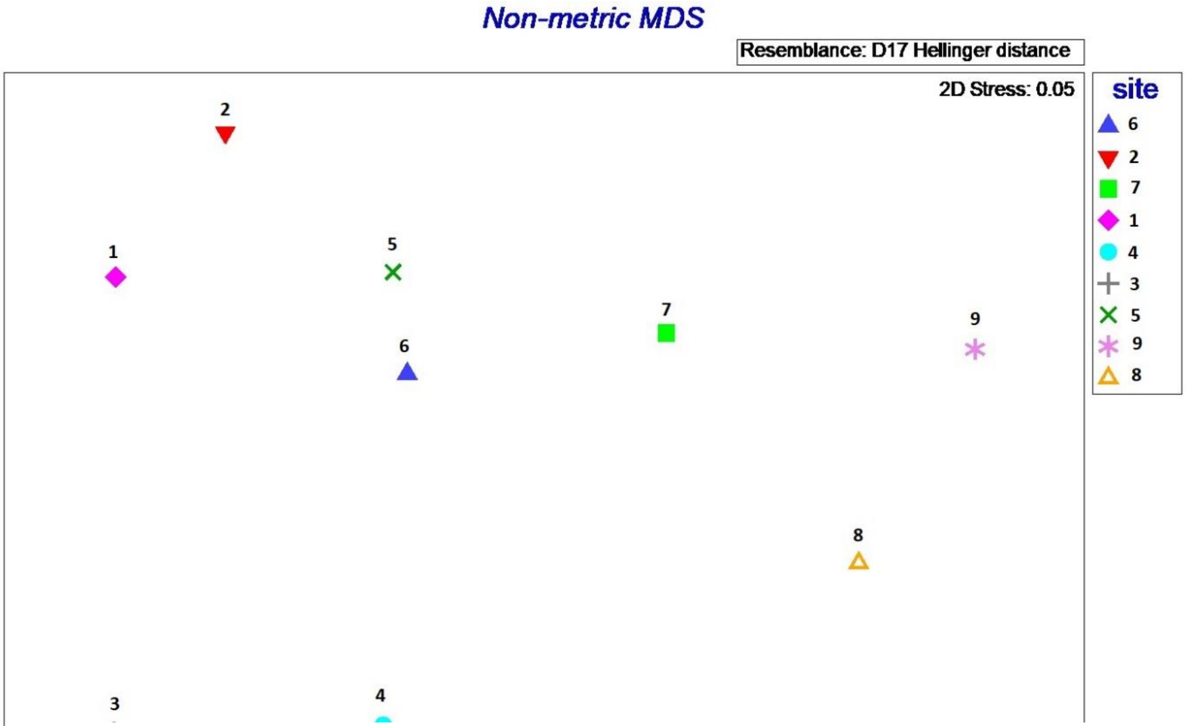


Figure 2.3. A plot of axis 1 versus axis 2 of an NMDS plot for the macroinvertebrate communities collected on nine deer farms in January 2021

2.1).

Table 2.1. Correlation coefficients between NMDS axes, biological and chemical variables measured at nine deer farms.

Variable	MDS1	MDS2
MCI	-0.95	0.18
QMCI	-0.50	0.78
ASPM	-0.92	0.36
Periphyton	-0.74	0.16
Sediment	0.69	-0.13
Temp	0.19	0.36
Riffle	-0.84	-0.05
Stream Health Check	-0.38	0.02
AN	-0.28	0.51
NNN	-0.03	0.79
TN	0.18	0.81
DRP	-0.16	-0.18
TP	0.07	0.30
SS	0.49	0.22
Segflow	-0.13	-0.80
Segslope	-0.18	0.41
Segshade	-0.15	0.27
Segripnative	-0.61	-0.23
Avgtemp	0.34	0.54
Daysrain	-0.28	-0.05
Avgslope	-0.57	-0.40
Width(m)	-0.22	-0.82
Depth(m)	0.13	-0.82
Velocity(m/s)	-0.54	-0.49

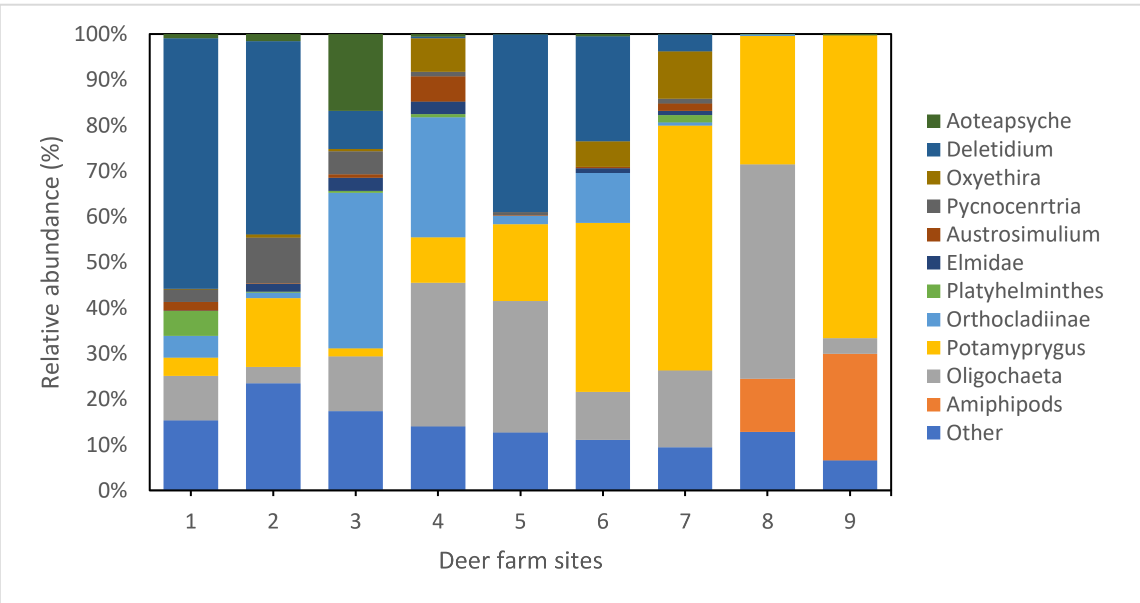


Figure 2.4. Relative abundance of macroinvertebrates at the nine deer farms in order from highest MCI score to lowest

Discussion

While the adverse impacts of dairy and sheep and beef farming on the ecological health of New Zealand waterways have been thoroughly investigated, there have been very few studies on the effects of deer farming (De Klein et al., 2003; Rodda et al., 2001). The recent changes to government policy on freshwater apply equally to all forms of animal agriculture. It is important to understand the drivers of ecological health on deer farms to ensure effective mitigation practices are implemented. The streams on these deer farms had biotic indices that indicated mild to severe impairment, with macroinvertebrate communities in streams with the highest levels of deposited sediment the most adversely affected. Water nitrogen concentrations had a mild impact on community composition but were not strongly linked with the biological indices implying that sediment rather than nutrients was the principal driver of stream health degradation.

The use of water chemistry monitoring has been adopted by a number of farm catchment care groups around New Zealand to assess the environmental effects of specific land use practices. These nine deer farms have had water chemistry sampled monthly for the last five years. However, the 2021 medians from this water chemistry monitoring were not linked with any measures of ecological health. Much of the research on the impacts of deer farming on freshwater has focused on the physical and chemical impacts of the deer without linking these to the ecological health of the stream through biological metrics. The research on water chemistry impacts of deer farming have implied increased nitrogen inputs, largely from urine and faeces in wallows connected to waterways, may be the driver of low ecological health on deer farm streams (Matthaei et al., 2006; McDowell, 2008, 2009a, 2009b; McDowell & Wilcock, 2008; Pattis et al., 2017).

However the total nitrogen concentrations on all my study streams is still lower than many dairy farm streams (McDowell & Wilcock, 2008; Quinn, 2000). The high nutrient levels associated with dairy farming have been linked to having a detrimental impact on macroinvertebrate communities (Ramezani et al., 2016; Townsend et al., 1997; A. Wright-Stow & R. Wilcock, 2017). The lack of a relationship between levels of any of the nutrients and the biological indices in this study suggests nutrient increases are not a water quality issue in deer farms. Furthermore, much of the current effort in water chemistry monitoring of these farms is probably unwarranted.

Reduced ecological health in streams with high levels of deposited sediment has been found in numerous studies (Broekhuizen et al., 2001; Burdon et al., 2013; Dolédec et al., 2006; Wagenhoff et al., 2011). Clearly high levels of deposited sediment are the primary impact of deer farming on waterway ecological health (Matthaei et al., 2006; McDowell & Wilcock, 2008; Rhodes et al., 2007). This is related to the wallowing and fence pacing behaviours of deer but also the degradation of riparian zones, bank trampling and soil compaction (De Klein et al., 2003; McDowell, 2008, 2009a; McDowell & Paton, 2004; Moore et al., 1985; Rodda et al., 2001; Sakai et al., 2013). Of the nine deer farming sites, three sites had deposited sediment cover below 50 percent. All nine sites allowed stock access to waterways, so the lower level of deposited sediment at these three sites was probably linked with their harder geology and higher quality riparian zones rather than deer exclusion from the stream.

The influence of geology on land use effects, such as the level of deposited sediment, has been highlighted in a number of studies (Quinn & Hickey, 1990a; Richards et al., 1993; Young et al., 2005). The type of geology present at the nine sites was likely influencing the degree to which the quality of the riparian zone and deer behaviour impacted increased deposited sediment levels and thus overall ecological health. While the interaction between geology and land use have been found to influence the health of a waterway directly, the riparian zone vegetation also interacts with these factors (Harding et al., 1998; Naiman & Decamps, 1997; Stauffer et al., 2000). The site with the highest levels of deposited sediment and the lowest ecological health had a soft sediment geology, a poor-quality riparian zone, and stags overwintering in the paddock. The combination of these three factors is likely to have cumulatively affected the severity of the bank erosion and sediment deposition and therefore the overall ecological health of the waterway.

The variation in macroinvertebrate assemblages varied across the nine sites reflected in the pattern seen in the biological indices, with communities dominated with pollution sensitive *Deletidium* mayflies in the low sediment streams. The sites with high sediment had lower index scores and communities dominated with pollution tolerant *P. antipodarum* and Oligochaeta. The sites with moderate periphyton growth and stream widths greater than 2 metres wide had communities with a lower abundance of Molluscs and higher abundance of Chironomidae. Streams in catchments with intensive agriculture have fewer pollution sensitive taxa such as Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (Harding & Winterbourn, 1995; Quinn, 2000; Quinn & Hickey, 1990a). Instead, communities are dominated by pollution tolerant taxa such as *Potamopyrgus*, *Oligochaeta* and *Chironomidae* (Hall et al., 2001; Harding & Winterbourn, 1995; Quinn & Hickey, 1990a).

As deposited sediment appears to be the principal driver of the ecological health in these streams, mitigation that reduces erosion will have the greatest impact on restoration (Rodda et al., 2001). It also suggests that deer farmers should be monitoring in-stream deposited sediment rather than nutrient levels to assess the ecological condition of their farm waterways. Stock exclusion and riparian afforestation have been found to reduce sediment deposition in streams and improve the ecological health of a stream (Greenwood et al., 2012; Parkyn, 2004; Quinn, 2000). In streams where deer have been excluded, deposited sediment levels have been seen to drop significantly over time and the macroinvertebrate communities increase in their relative abundance of pollution sensitive taxa (McDowell, 2009a; Sakai et al., 2013; Sakai et al., 2012). Sakai et al. (2012), described wild Sikar deer in Japan as being ecosystem engineers. The wild deer were able to influence the macroinvertebrate communities of streams through the degradation of riparian vegetation, stream banks and the stream channel. Areas where deer had been excluded saw reduced deposited sediment, the recovery of riparian vegetation and increased taxa abundance over a four year period (Sakai et al., 2013).

Fencing waterways alone may not always address the sediment and nutrient input from wallows. McDowell (2009a, 2009b), suggested that wallows connected to waterways and wetland areas should also be fenced off and planted. Fenced off wallows should be replaced with the formation of artificial wallows not connected to waterways (McDowell et al., 2008). This allows deer to exhibit natural behaviours while minimising the environmental impacts. The use of unconnected artificial wallows in conjunction with fencing and planting has been found to result in a greater reduction in sediment and nutrient loading in deer farming streams than fencing and planting alone (McDowell, 2006, 2009b).

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Appendix 2.1

Correlation coefficients for biological metrics and water chemistry, periphyton biomass

	AN	**NNN**	**TN**	**DRP**	**TP**	**Periphyton Biomass**
MCI	0.23	0.12	-0.05	0.30	0.15	0.56
QMCI	0.52	0.59	0.54	0.21	0.39	0.40
ASPM	0.45	0.32	0.14	0.19	0.09	0.65

Chapter 3

Does the Stream Health Check correlate to the ecological health of a waterway?

Abstract

The Stream Health Check was a tool developed to help landowners assess the quality of their waterways while also providing education and promoting environmental awareness. It aims to help landowners identify areas that require prioritisation for freshwater management. The purpose of this study was to investigate whether the score from a Stream Health Check assessment correlated with freshwater macroinvertebrate biological metrics of the site. The scores from the Stream Health Check correlated with the MCI of deer farm streams. It also correlated well to both the MCI and the QMCI of streams in mixed agricultural practice catchments. This suggests that the Stream Health Check would be a useful tool for landowners in the development of Environmental Farm Plans as it is an inexpensive way to assess the physical and ecological quality of a waterway that relates well to more scientific methods for assessing ecological health.

Introduction

Animal agricultural can have detrimental impacts on freshwater environments (Quinn, 2000; Vörösmarty et al., 2010). Recent governmental policy reform in freshwater, along with changing public expectations, are placing greater pressure on the animal agriculture industry to become more environmentally sustainable (Ministry for the Environment, 2020; Stokes et al., 2021). There has been a focus on stock exclusion and riparian planting with the intention of reinstating a vegetated riparian margin to improve the water quality and ecological health of waterways (Collins et al., 2013; Dosskey et al., 2010; Perry et al., 1999).

The riparian margin is an important habitat that facilitates the ecological health of a waterway (Greenwood et al., 2012; Parkyn, 2004; Quinn et al., 1993; Quinn, 2000). Damage to the riparian margin is common in animal agriculture particularly if stock have direct access the waterway (Quinn, 2000). Restoration of riparian margins has been found to reduce nutrient input and increase shading of the water (Dosskey et al., 2010; Parkyn, 2004). This can reduce periphyton biomass and water temperature (Rutherford & Cuddy, 2005). Riparian planting has been found to stabilise stream banks and reduce erosion and thus lower deposited sediment infiltration (Greenwood et al., 2012; Wilcock et al., 2009). Riparian planting is often used in conjunction with stock exclusion which will also reduce sediment and nutrient inputs by preventing bank trampling, vegetation damage and direct nutrient input from urine and faeces (Collins et al., 2007; Parkyn, 2004). Riparian zones also impact the habitat quality and organic resources supply to instream life (Broekhuizen et al., 2001; Liess et al., 2012).

As the quality of the riparian margin has large influences on the environmental state of a waterway, it will also affect freshwater macroinvertebrate communities (Jowett et al., 2009; Naiman & Decamps, 1997). Sites with low quality riparian zones have been found to have greater proportions of pollution tolerant taxa such Diptera, Mollusca and Oligochaetes and reduced proportions of Ephemeroptera (Reid et al., 2010). The MCI and QMCI scores of waterways with high quality riparian zones are also higher than those of streams with degraded riparian zones (Collier et al., 1998).

The management of the riparian margin is largely the responsibility of the landowner. A recent survey of deer farmers found that while most farmers had a moderate level of environmental awareness, they struggled to identify specific environmental issues and steps to minimise their environmental impact (Payne & White, 2006). The waterway self-assessment form was developed as a tool for land owners to assess the state of their waterways and provide indirect education on freshwater health and riparian management (Polglase, 2000). This assessment form allows farmers to assess their waterway using a range of visual assessments without the need for any specific equipment. The assessment form has a series of questions with potential answers for each specific waterway fitting into one of four possible categories, each worsening state receiving concomitantly lower scores. The scores are summed for all the questions to give an overall metric. The lower scores indicate that mitigation is necessary to improve the quality of the riparian zone, waterway habitat or ecological health.

In this study we have modified the waterway self-assessment form developed by (Polglase, 2000) and renamed it as the Stream Health Check (Appendix 3.1). We sampled several waterways in the North and South Island and compared the stream health check assessments to biological metrics based on instream invertebrate sampling and collections of water samples for chemical assessment. We used this comparison to assess whether the stream health check score is a valid indication of the ecological health of a waterway influenced by animal agriculture.

Materials and Methods

Stream Health Check

A modified version of the waterway self-assessment form (Polglase, 2000) was developed for use on sheep and beef farms as well as deer farms. This form is a tool that can be used by researchers and landowners to get a gain a holistic understanding of the existing characteristics of a waterway. The form is currently being used by Beef and Lamb to form a portion of their farm plans that can be written by landowners (Beef + Lamb NZ, 2021) . The stream health check scores a site on four categories; the stream bank, instream life, potential for contaminants and the stream channel. The score generated by the assessment can then be used to indicate variables that could be improved and therefore require mitigation (Appendix 3.1).

Sites

Nine high country and hill country deer farms across New Zealand were assessed using the stream health check. The nine sampled sites are deer farms in the Hawkes Bay, Waikato, Marlborough, Tasman, Otago and Southland regions (Figure 3.1).

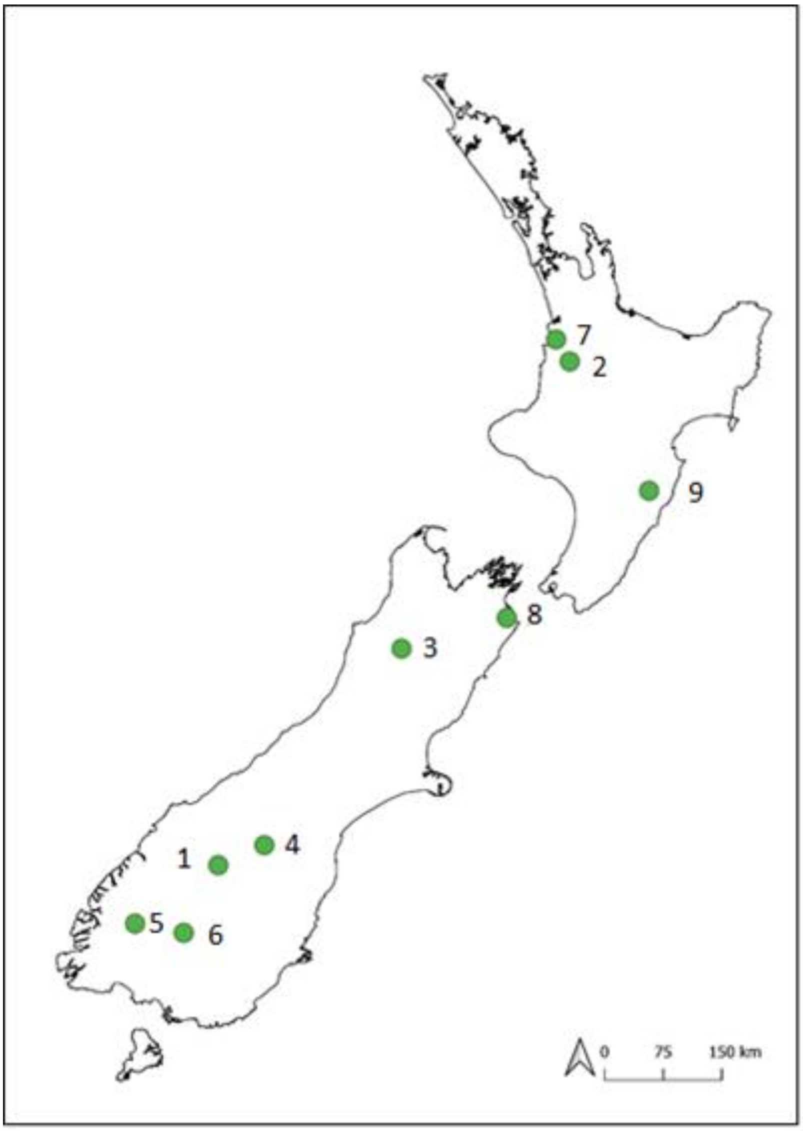

Figure 3.1 The nine deer farming sites around New Zealand

Site 1 is a 2000-hectare block of tussock dominant high country in Otago. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above between November and February of each fawning season. The site had historically been used for gold mining. Site 2 is a 770-hectare block of rolling hill country in Waikato. The sampling site had no stock exclusion fencing with beef cattle and deer in the catchment above the site. The site had evidence of bank tramping. The stream was a stony bottomed. Site 3 is a 440-

hectare block of river flats and terraces in Tasman. The sampling site had no stock exclusion fencing with breeding stock in the catchment above. Site 4 is a 3500-hectare block of medium high country in Otago. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above. The site had some bank erosion present due to slips. Site 5 is a 5500-hectare block of tussock dominant hill country in Southland. The sampling site had no stock exclusion fencing and the catchment above is used for fawning. Site 6 is a 1570-hectare block of steep to medium hill country in Southland. The sampling site had no stock exclusion fencing with hinds and fawns in the catchment above. The site had schools of Gollum Galaxiid present. Site 7 is a 980-hectare block of steep hill country in Waikato. The sampling site had no stock exclusion fencing and had a mix of livestock classes. This catchment represents the highest rainfall catchment of the 9 sites. Koura were also present during invertebrate sampling. Site 8 is a 180-hectare block of flat to rolling hill country in Marlborough. The sampling site had stags wintering in the paddock with no stock exclusion fencing. The site had substantial bank erosion and slow flowing water with reaches of the stream drying up fully in summer. Site 9 is a 550-hectare block of steep hills to rolling country in Hawkes Bay. The sampling site had no stock exclusion fencing with only deer in the catchment above the site. The site had evidence of bank tramping and high levels of deposited sediment. Site 7 was removed from the data set as it had had significant mitigation practices that had been introduced in the 12 months prior meaning that it scored well on the stream health check. The biological metrics for Site 7 were still low though as the ecological health of the waterway had not had sufficient time to react.

Nine sites in the Pohangina Valley for which Stream Health Check and invertebrate data were available were also used. Some of the sites had deer farming in the catchment above the waterways but most were in sheep and beef agriculture.

Invertebrate sampling

Invertebrate samples from Chapter 2 sites were collected between January and March, 2021. Macroinvertebrates were collected from riffles and runs within a 100m reach of the stream using a 0.5mm mesh Surber sampler. At each site five Surber samples were collected and preserved in 90% ethanol. The samples were processed in the laboratory using a 200 individual fixed count with scan for rare taxa. Invertebrates were identified to genus level using Winterbourn et al. (1989). A range of biological indexes were calculated from each sample, including the MCI (Macroinvertebrate Community Index), the QMCI (quantitative MCI) and ASPM (Macroinvertebrate Average Score Per Metric) (Clapcott et al., 2017; Collier, 2008; Stark & Maxted, 2007). The taxa tolerance scores described in Stark and Maxted (2007); Winterbourn et al. (1989) were used for calculating the MCI and QMCI scores. When normalising the data for the ASPM calculations, the national maximum and minimum were applied (Environment, 2020).

Additional invertebrate samples from nine waterways in the Pohangina Valley, North Island, were collected in an identical way to above between March and May 2021. The samples were processed in a similar way except that full counts rather than fixed counts were used for enumeration.

Data analysis

Linear regression models were used to investigate whether Stream Health Check scores were related to biological metrics calculated on invertebrate samples using Rx64 4.0.2 (R Core Team, 2022).

Results

High country and hill country deer farms

Of the nine high country and hill country deer farms, 8 scored above 120 in the stream health check indicating an intermediate level of health. Of the nine high country and hill country sites 5 scored below the national bottom line for MCI scores with 2 below the national bottom line for ASPM scores. Most of the sites had MCI scores that indicated mild to moderate organic pollution and eutrophication. The MCI were higher at sites that had also scored higher on the stream health check ($F_{1,43}$ = 20.83, P<0.001, r^2 = 0.34) (Fig. 3.2). There was no correlation between the Stream Health Check score and the QMCI score, nitrate or DRP levels.

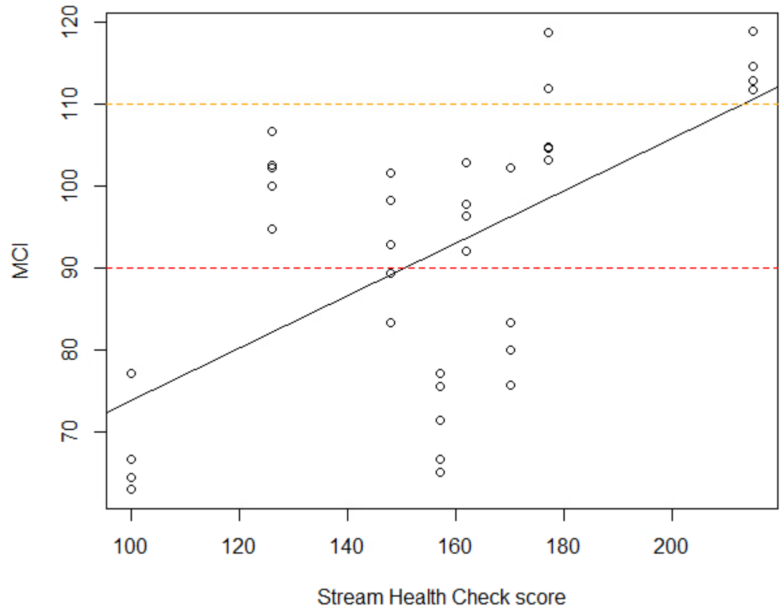

Figure 3.2. Plot of MCI as a function of the Stream Health Check score collected at 9 deer farm streams in 2021. Data points are Surber samples. The red dotted line indicates the national bottom line set in the NSPFM, the yellow dotted line indicates the lower limit for class B.

Pohangina Valley Farms

The nine Pohangina valley sites scored higher on the stream health check as well as the MCI scores. Six of the sites scored above 250 indicating that these sites would be low priority for waterway management. All nine sites had MCI scores above the national bottom line. The MCI and QMCI was higher at sites that had also scored higher on the stream health check ($F_{1,43}$ = 38.31, P<0.001, r^2 = 0.46, $F_{1,43}$ = 11.67, P<0.01, r^2 = 0.19, MCI and QMCI respectively) (Fig.3.3). The dissolved reactive phosphorus at the Pohangina Valley sites did not correlate with the Stream Health Check score but sites with higher Stream Health Check scores did have high nitrate levels ($F_{1,7}$ = 6.08, p<0.05, r^2 = 0.39).

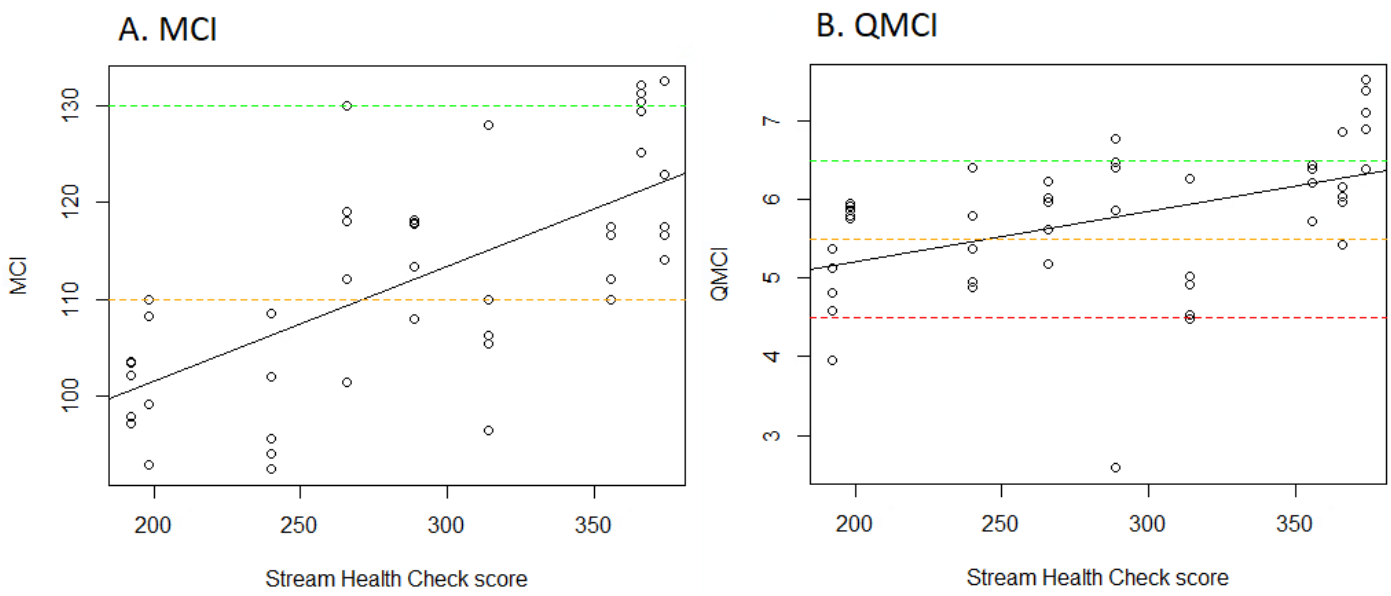

Figure 3.3. Plot of A. MCI and B. QMCI as a function of the Stream Health Check score collected at nine sites in the Pohangina Valley. The red dotted line indicates the national bottom line set in the NSPFM, the yellow dotted line indicates the lower limit for class B and the green dotted line indicates the lower limit for class A.

Discussion

The stream health check scores for the deer farms and the Pohangina Valley farms correlate with MCI scores for the sites. Polglase (2000) also found a strong link between the forerunner of the Stream Health Check (Waterway Self-Assessment Form) and MCI at 30 sites throughout the Manawatu and Rangitikei. The stream health check did not relate to the quantitative version of the MCI, the QMCI, as strongly. In fact, for the high and hill country deer farms, the QMCI score was not correlated with the stream health check scores even though the MCI was. Both metrics correlated with the stream health check scores at the Pohangina Valley sites. The Pohangina Valley sites had higher biological metric scores than the deer farms as these sites are in an area of relatively low intensity agriculture with streams feed from native forest headwaters.

The exclusion of the deer farm Site 7 during the data analysis highlights circumstances where the Stream Health Check may not correlate well with the biological metrics measured at a site. Farms with recently implemented mitigation may find that these practices have not had sufficient time to influence instream communities and the overall ecological health of a waterway. There are considerable lag times between physical habitat improvement and ecological reinstatement of a waterway (Harding et al., 1998). The overall effectiveness and the time needed for ecological improvement can be even further delayed if the mitigation practices are not catchment wide (Greenwood et al., 2012; Harding et al., 1998). Site 7 was in the process of fencing the waterway, riparian planting, and the establishment of sediment traps higher in the catchment. This meant that the site scored high on the Stream Health Check despite the biological metrics indicating habitat degradation. The monitoring of the effectiveness of mitigation practices implemented also needs to take into consideration the amount of time required for reestablishment of invertebrate communities before concluding whether a mitigation was a success.

The Stream Health Check provides a simple and effective tool for farmers to assess their waterways that correlates to the ecological health. It is also an effective way for farmers to identify specific areas that require improvement and help improve awareness of farming activities that may adversely impact waterway health (Polglase, 2000). The future use of this tool for the development of freshwater farm plans could be useful. It provides farmers with an inexpensive yet effective tool for self-driven environmental awareness while providing guidance for environmental improvement. A review of Canada's approach to addressing agro-environmental management found that consistent scientific engagement with farmers can increase the success of sustainable agriculture projects (Kröbel et al., 2021). By scientists collaborating with farmers and providing them with the knowledge and skills to drive environmental sustainability at a farm-by-farm level there is a greater level of personal responsibility and overall better environmental outcomes (Kröbel et al., 2021). There is growing awareness among New Zealand farmers that environmental sustainability needs to be considered as New Zealand agriculture faces both legislative and socially driven changes (Fairweather & Keating, 1994; Michel-Guillou & Moser, 2006). The Stream Health Check is a tool that helps to indirectly educate farmers around the health of their waterways and help them identify which factors require prioritisation for mitigation practices.

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Appendix 3.1

Stream Name			**Date &Time**	
1. Stream banks				
1a. What type of vegetation is along the banks stream side?	Trees with dense groundcover e.g. tussock, toetoe, ferns, flax rushes.	Tall grasses with patchy trees and groundcover.	Patchy trees, groundcover grazed or absent.	Grazed pasture grasses to stream edge.
	16	8	4	2
1b. How continuous is the vegetation (other than pasture) along the stream banks?	Tall vegetation/trees continuous, or a few small gaps.	Tall vegetation/trees less continuous, a few large gaps or several small gaps.	Breaks in tall vegetation/trees frequent - very patchy.	Many large gaps in tall vegetation/trees or no tall vegetation at all
	16	8	4	2
1c. What is the average width of the vegetation (other than pasture) along the stream banks?	> 30 m	10 - 30 m	1 - 10 m	< 1 m
	32	16	8	4
1d. What percentage of the stream is shaded by plants and stream banks?	50% or more.	30%.	10%.	Little or no shading.
	16	8	4	2

1e. How stable are the stream banks?	Banks stable, rock and soil firmly held by grasses, shrubs, and tree roots.	Banks firm but loosely held by grass and shrubs.	Banks of loose soil held by a patchy layer of grass and shrubs.	Banks unstable, of loose soil or sand easily disturbed.
	16	8	4	2
1f. What is the level of erosion on surrounding landscape and on the stream banks?	No evidence of erosion in surrounding land area, no scarring on stream banks and no undercutting.	Some erosion in surrounding land. Occasional scarring on stream banks and undercutting.	Moderate erosion in surrounding land. Eroding banks slowly widening.	Significant erosion in surrounding land area. Significant areas of stream bank cut away, some loss of farmland.
	32	16	8	4
2. Instream life				
2a. What is the level of periphyton (algae) slime? **NB: This needs to be assessed in summer AND after about two weeks without any flooding**	Stones rough to the touch Scraping thumb nail over stones yields no slime.	Stone slippery to touch Scraping thumb nail over stones yields no slime.	Stone very slippery to touch Scraping thumb nail over stones yields a small about of slime.	Thick layers of slimy algae Scraping thumb nail over stones yields large volume of slime.
	32	16	8	2
2b. Are there any natural obstructions to stream flow?	Rocks and old logs firmly set in place.	Rocks and logs back filled with sediment.	Rocks and logs lose, moving with floods.	No obstructions to slow stream flow.
	16	8	4	2

2d. What invertebrates (bugs) are present in your stream? **NB: To find stream insects look under rocks. Or if the stream has no rocks on water weeds, grass, logs & other debris**	Lots of mayflies, stoneflies and other types of crawling and swimming insects.	Moderate numbers of mayflies and caddisflies. Variety of other types of insect may also be found.	Very few crawling and swimming insects. Snails, worms and midges abundant.	Mostly snails, worms and midges
	32	16	4	2
2e. How often does your stream overtop (overflow) its banks?	Never known to overtop banks.	Overbank flows rare.	Overbank flows occur during some winter storms.	Overbank flows frequent in winter/spring storms. Or stream channelised.
	16	8	4	2
3. Potential for contaminates				
3a. Do stock have access to your stream?	Stock do not have access to any of the stream or its banks.	Cattle or deer have access to only a small part of the stream. Sheep have access.	Cattle, deer or sheep have access to most of the stream.	Stock have access to the entire stream.
	32	16	8	4

3b. What is the potential for the input of sediment to your stream? (e.g., from stream banks, stock damage/trampling, stock crossings, surface runoff, runoff from farm roads, slips/erosion, gravel extraction, etc.).	No potential.	Low potential.	Moderate potential.	High potential.
	32	16	8	4
3c. What is the potential for the input of contaminants to your stream? (e.g., from spray drift, sprayer washings sheep dips, effluent ponds, silage pits, dumps, oil and foam, dead animals, etc.)	No contamination.	Low contamination.	Moderate contamination.	High contamination.
	32	16	8	4

3d. Is there any artificial drainage entering the stream? (e.g., tile, mole, storm water, and/or open drainpipes which have vegetation dredged).	No artificial drainage.	Sparse artificial drainage.	Moderate amount of drainage.	Extensive drainage networks.
	32	16	8	4
3e. Are there any Critical Source Areas (CSA's) or overland flow pathways where runoff enters the stream (e.g. gullies, depressions, swales on adjoining land).	No CSA's or overland flow pathways within100 m.	One CSA's or overland flow pathways within100 m.	2-3 CSA's or overland flow pathways within100 m.	Greater than 3 CSA's or overland flow pathways within100 m.
	16	8	4	2
3f. How much nitrogen and phosphorous fertiliser do you use annually	None	Less than 150 kg/ha super or equivalent, no nitrogen fertiliser.	150 - 300 kg/ha super or equivalent, less than 50 kgN/ha.	More than 300 kg/ha super or equivalent, greater than 50 kgN/ha.
	32	16	8	4
4. Stream channel				

4a. How steep and deep are your stream banks?	Top of stream banks 10 m or higher above stream.	Top of stream banks 10 - 5 m above stream.	Top of stream banks 5 - 1 m above stream.	Top of stream banks 1 m or less above stream.
	16	8	4	2
4b. What is on the streambed?	Rocks and stones of different sizes, tightly packed together.	Stones, silt present in gaps between rocks/stones.	Gravel, sand, and silt.	Sand and silt, stones absent.
	16	8	4	2
4c. If you stand in the stream and dig your feet into the substrate does the water, go murky	Remain clear	Clear quickly	Remain murky for less than 1 minute	Remain murky
	32	16	8	4
4d. How cohesive are the soils of the stream bank?	Very cohesive. Mostly rock and cemented material (boulders and bedrock).	Reasonably cohesive. Tightly packed gravel or sand in a clayey matrix.	Loose soils with fine aggregates. Tightly packed sands or gravel with some silt or clay.	Very loose soils. Loosely packed sandy, gravelly, or pumice material.
	16	8	4	2
4e. How well do your soils drain after rain?	Deep, well-draining soils that slow down the recharge of water to waterways and drains.	Moderately well-draining soils, but soils are waterlogged for long periods in winter.	Excessively well-draining soils - water moves freely through the soil, reaching waterways rapidly, sandy/porous soils.	Poorly draining soils. Soils water-logged for long periods after rain, surface ponding occurs in low areas. Heavy textured soils.
	16	8	4	2
TOTAL (add up all the scores to generate a total)				