THE EFFECT OF LANDUSE AND GEOLOGY ON MACROINVERTERBATE COMMUNITIES IN EAST COAST STREAMS, GISBORNE, NEW ZEALAND.

A stream draining exotic forestry flowing into pastoral land use, in soft geology, Gisborne, New Zealand.

THE EFFECT OF LANDUSE AND GEOLOGY ON MACROINVERTERBATE COMMUNITIES IN EAST COAST STREAMS, GISBORNE, NEW ZEALAND.

Harriet Roil

Masters Thesis

Massey University

March 2020

Table of Contents

Acknowledgements

Firstly I would like to thank my Husband, Liam for his support and encouragement over the last two years while I have worked and studied. Completing this wouldn't have been possible without your support and thanks for ensuring I have been kept fed and watered.

I would like to thank my supervisors. Russell Death, Professor of Freshwater Ecology at Massey University, for allowing me to complete my Masters extramurally and the support you have given to make this happen. I genuinely appreciate the effort you have made for this. The guidance you have given throughout this thesis has been invaluable and has calmed stressful times. Your assistance with statistics and editing has been appreciated. Russell, thank you. Ian Fuller, Professor in Physical Geography at Massey University, my second supervisor, thanks for your support and the endless questions you have assisted me with in all things geology and geomorphology.

Thanks to Dimitrios Rados from Massey University for your assistance with the processing of macroinvertebrate samples, I genuinely could not have done this without your support, so thanks. I would also like to thank Anja Moebus who assisted with the analysis of sediment particle sizes.

I would like to thank Ernslaw One Limited for allowing me to conduct my research within their forest. A special thanks to James Sinclair and Billy Maaka who provided invaluable knowledge about sites, and for showing me around the forest and assisting with questions.

Gisborne District Council provided data and support for sampling and I would like to especially thank Lois Easton and Paul Murphy for their support in giving me time to collect data and their support throughout the process. I would like to thank Peter Hancock for providing the dataset for the State of Environment data.

Thanks to my work colleagues for putting up with me when I was short for time, especially a big thanks to Abigail Salmond who has not only provided advice and listened to my challenges but has supported me in my work-study life.

I would finally like to thank my family, for reminding me of why I started this process and for always been there when I have needed times of sanity.

General Introduction

Native forests once dominated the land cover in New Zealand and following colonisation and the clearance of forests from both Maori and European people there has been a reduction in indigenous land cover from 90 to 24% (McGlone, 1989). Lowland areas with fertile flats and low gradients have been decimated of indigenous forest due to their conversion to pasture and horticulture for productive, intensive land use and urban development (McGlone,1989). Currently the predominant areas of indigenous forest in New Zealand are in conservation reserves controlled by the Department of Conservation and indigenous forest is fragmented around both the North and South Islands (Ewers *et al*., 2006).

New Zealand land use is now dominated by primary production including; agricultural (40%) hill country, extensive farming systems and dairy farms, production forestry (8%), horticulture (2%) and industrial and urban centres (0.8%) (Stats NZ, 2018). These have replaced the original landscape consisting of mixed indigenous forests, wetlands and tussock grasslands which all formed an important role in freshwater ecosystem function (Ewers *et al*., 2006, MacLeod and Moller, 2006). The changes in land use have put pressure on the water resource and the balance of maintaining economics and freshwater ecosystem integrity are a continual struggle for resource managers (Shearer and Young, 2011).

The scale and intensity of land use within catchments influence freshwater ecosystem function and structure and a number of studies have explained the effects of land use on freshwater communities (Quinn and Hickey 1990; Death *et al*., 2003; Parkyn *et al*., 2006; Thompson *et al*., 2009; Howard –Williams *et al*., 2011). Land use change in New Zealand has caused changes in hydrological processes, flow and thermal regimes, radically modified stream morphology, degraded water quality, increased nutrients, reduced woody debris and inputs of leaf litter and increased sediment loads (Quinn *et al*., 1997; Quinn and Hickey, 1990; Newson, 2000; Death *et al*., 2003; Clapcott *et al*., 2011; Howard –Williams *et al*., 2011). The shifts in vegetation and intensity of land use can therefore cause fundamental changes in habitat structure and quality for freshwater fauna (Death *et al*., 2003; Parkyn *et al*., 2006), creating favourable habitat for sediment tolerant invertebrates such as Elmidae, Chironomidae and Oligochaeta (Death *et al*., 2003), providing more food for grazing invertebrates and shifting the community assemblage from being dominated by Ephemeroptera, Trichoptera and Plecoptera species to those dominated by Diptera, Molluscs and Crustaceans (Quinn and Hickey 1990; Quinn *et al*., 2009).

The percentage of pastoral land use in a catchment influences a wide range of aspects of stream health (Quinn and Hickey 1990). The introduction of contaminants from agriculture are relatively well known (Mcdowell *et al*., 2004). Diffuse and point source discharge occurs (with the movement of these nutrients often creating time lags in contamination), or inputs of sediment and faecal matter directly enter waterways via overland flow or via stock access to waterways (McDowell and Wilcock, 2008; Howard-Williams *et al*., 2011; Julian *et al*., 2017). Intensive land use has higher stocking rates, fertiliser inputs and production outputs and slope and soil type can determine the level of nutrient and sediment runoff resulting in increased, turbidity, sediment and nutrient levels in waterways (Howard –Williams *et al*., 2011; Julian *et al*., 2017). Dairy farming has the highest diffuse pollution footprint compared to other agricultural farming practices with 37% of the total N load originating from 7% of the land area occupied by dairy farms (McDowell and Wilcock, 2008; Howard-Williams *et al*., 2011). However the negative effects on water quality from beef cattle can be similar with grazing on steep land, requirements for good pasture and continual developments in modern pasture management (Julian *et al*., 2017). Pastoral land use increases water yields in catchments via a reduction in forested area and increased surface run off, however in intensive irrigated areas catchment water yields are decreased through water abstraction and evapotranspiration (Schmidt *et al*., 2009). Increased contaminants through surface run off are worse in agriculture compared to those in exotic or indigenous forested catchments (Death *et al*., 2003; McDowell and Wilcock, 2008).

Plantation forestry in New Zealand is dominated by *Pinus radiata* and includes a rotational cycle (establishment, growth, harvest, replanting) of 28-30 years (Quinn and Phillips, 2016). Plantation forests can improve water quality from pastoral land use and can have similar stream health to that of indigenous forest (Death *et al*., 2003; Quinn *et al*., 2009). At the catchment scale clear-fell harvest can alter stream flow characteristics and generate higher peak flows and disturbance (Quinn *et al.,* 2009; Thompson *et al.,* 2009). Disturbance includes the exposure of large areas of bare soil, movement of large quantities of sediment, and the transportation of nutrients and organic material into streams following rainfall or storms (Marden *et al*., 2006). These effects coupled with the removal of bank side vegetation, changes to thermal regimes, increases in light and temperature, and changes in stream morphology and flow regimes have been shown to influence macroinvertebrate diversity and richness (Death *et al.,* 2003; Quinn and Wright-Stow 2008). Following forestry harvest, macroinvertebrate communities resemble the same composition to those found in pastoral streams (Quinn, 2000; Death *et al*., 2003).

The increased levels of deposited sediment (inorganic particles <2mm) through land use change has had some of the most far reaching consequences on freshwater ecosystems and biological processors (Clapcott *et al*., 2011). Sediment can influence benthic communities including changes in feeding growth, behaviour, community composition, diversity and abundance and alterations to fish habitats and diets (Clapcott *et al*., 2011). Levels of deposited sediment in indigenous forest are generally relatively low and therefore the adverse effects of fine sediment on freshwater biota can occur following small increases of sediment cover (Wagenhofff *et al*., 2013). The accelerated anthropogenic change that has resulted in increased sediment levels means species have not had time to adapt or change to the conditions, and therefore sediment drives a change in species composition from intolerant to tolerant taxa (Death *et al*., 2003; Hobbs *et al*., 2006; Clapcott; *et al*., 2011).

Increased international demand for soft wood and the contribution of trees in carbon sequestration commitments has resulted in an intensification of both plantation and indigenous forests in New Zealand (Carnus *et al.*, 2006). Investment in new plantation forest in New Zealand currently is high, with the establishment of the 1 Billion Trees project in 2018 encouraging areas to be planted in forestry. Both indigenous and exotic trees have been planted (58% and 42% respectively) with the goal that one billion trees are planted by 2028, predominantly on private land with nearly five million trees having been planted at the end of 2019 (Te Uru Rakau 2019). Large areas of agricultural land have been purchased for afforestation under this scheme with the intention that some of these forests will have a rotational cycle and be harvested. Although water quality may be improved through the conversion of pasture to forest (Death *et al*., 2003; Fahey *et al*., 2003), increasing forestry growth is coupled with a requirement to understand the potential negative influences on freshwater ecosystems from the rotational cycle (Quinn and Phillips, 2016). The environmental stressors influencing communities following harvest are well known (Death *et al.*, 2003; Parkyn *et al.*, 2006; Quinn and Wright-Stow, 2008; Quinn *et al.*, 2009), however regional influences such as geology and climate, could be equally important in determining the response of freshwater ecosystems (Shearer and Young, 2011).

Geology, climate and vegetation type can drive sediment distribution, water chemistry and hydrology at catchment scales (Allan and Johnson, 1997; Richards *et al*., 1997; Shearer and Young, 2011). Human activities, changes to landscapes and land uses have been the fourth controlling factor on freshwater ecosystem services and function (Townsend *et al*., 1997). Magnifications of impacts on ecosystems from land use may differ depending on underlying geology, climates and catchment vegetation as all these factors interact over both space and time (Allan and Johnson, 1997; Shearer and Young, 2011; Julian *et al*., 2017). Therefore influences of land use on freshwater communities are complex and could be regionally specific because of unique combinations of geographic location, geology, topography, climate and altitude (Biggs *et al*., 1990, Julian *et al*., 2017). Parkyn *et al*., (2006) found that impacts on water quality from land use in soft geology were different compared to other locations of New Zealand and Shearer and Young, (2011) demonstrated macroinvertebrate communities in exotic forest were similar to indigenous forest communities in granite geology but not underlying gravel geology.

The last decade in New Zealand has put focus on the ability for freshwater systems to maximise ecosystem services provided by healthy rivers through legislation such as the National Policy Statement for Freshwater Management (2014). With increasing diversification and intensification of land use in New Zealand, it is important to understand the link between geology, climate and land use on freshwater communities. The understanding of spatial and temporal relationships with land use will assist with appropriate management decisions to achieve the goals of the National Policy Statement for Freshwater Management (2014) and to improve freshwater ecosystems in New Zealand rivers. Few studies have been undertaken on land use and freshwater ecosystem interactions in different land use and geology types. With time since disturbances such as forest harvest, community population dynamics and organisation influenced by not only the activity, but also environmental stressors exacerbating issues to freshwater communities. The following two chapters address the interactions between land use, macroinvertebrate communities and environmental stressors in the East Coast region of the North Island, New Zealand. Chapter one explores the interactions between land use and geology and macroinvertebrate communities in the East Coast region of the North Island. Chapter two explores the recovery time of macroinvertebrate communities in soft geology following forestry harvest.

References

Carnus, J.M., Parrotta, J., Brockerhoff, E., Arbez, M., Jactel, H., Kremer, A., Lamb, D., O’Hara, K., Walters, B., (2006). Planted forests and biodiversity. Journal of Forestry 104, 65–77.

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

Close, M., Dann,R., Ball,A., Pirie,R., Savill,M., Smith,Z., (2008). Microbial groundwater quality and its health implications for a border-strip irrigated dairy farm catchment, South Island, New Zealand. *J Water Health* 1 March 2008; 6 (1): 83–98. doi: https://doi.org/10.2166/wh.2007.020

Death, R., Baillie, B., and Fransen, P. (2003). Effect of Pinus radiata logging on stream invertebrate communities in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research. 37. 507-520. 10.1080/00288330.2003.9517185.

Ewers, R. M., A. D. Kliskey, S. Walker, D. Rutledge, J. S. Harding, and R. K. Didham. (2006). Past and future trajectories of forest loss in New Zealand. Biological Conservation 133:312-325.

Fahey, B., Marden, M., & Phillips, C. (2003). Sediment yields from plantation forestry and pastoral farming, coastal Hawke's Bay, North Island, New Zealand. *Journal of Hydrology (New Zealand), 42*(1), 27-38. Retrieved February 21, 2020, from www.jstor.org/stable/43944858

Hobbs, R. J., S. Arico, J. Aronson, J. S. Baron, P. Bridgewater, V. A. Cramer, P. R. Epstein, J. J. Ewel, C. A. Klink, A. E. Lugo, D. Norton, D. Ojima, D. M. Richardson, E. W. Sanderson, F. Valladares, M. Vila, R. Zamora, and M. Zobel. (2006). Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15:1-7.

Howard-Williams, C., Davies-Colley, R., Rutherford, K., Wilcock, R. (2011). Diffuse pollution and freshwater degradation: New Zealand approaches to solutions. Pp 126-140. In: van Bochove, E; Vanrolleghem, P.A.; Chambers, P.A.; Thériault, G.; Novotná, B. and Burkart, M.R. (eds.). Issues and Solutions to Diffuse Pollution: Selected Papers from the 14th International Conference of the IWA Diffuse Pollution Specialist Group, DIPCON 2010. Conférence sur la pollution diffuse 2010 inc., Québec, Québec, Canada.

Julian, J. P., de Beurs, K.M., Owsley,B., Davies-Colley,R.J., and Ausseil, A.G.E., (2017). River water quality changes in New Zealand over 26 years: response to land use intensity. Hydrology and Earth System Sciences 21:1149-1171.

MacLeod, C. J., and H. Moller. (2006). Intensification and diversification of New Zealand agriculture since 1960: An evaluation of current indicators of land use change. Agriculture, Ecosystems and Environment. 115:201-218.

Mcdowell, R., and Wilcock, R.J. (2009). Water Quality and the Effects of Different Pastoral Animals. New Zealand veterinary journal. 56. 289-96.

McGlone, M. (1989). The Polynesian settlement of New Zealand in relation to environmental and biotic changes:New Zealand Journal of Ecology 12:116-129.

Newson, M. D., and. Newson. C. L. (2000). Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. . Progress in Physical Geography 24:1195- 1217.

Parkyn, S.M.; Davies-Colley, R.; Scarsbrook, M.R.; Halliday, N.J.; Nagels, J.W.; Marden, M.; Rowan, D. (2006). Pine afforestation and stream health: a comparison of land-use in two soft rock catchments, East Cape, New Zealand. New Zealand Natural Sciences 31: 113-135.

Quinn, J.A., Cooper,B.C., Davies-Colley, R.J., Rutherford,C.J., and Williamson, B.R., (1997) Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams, New Zealand Journal of Marine and Freshwater Research, 31:5, 579-597, DOI: 10.1080/00288330.1997.9516791

Quinn, J.M., Croker,G.F., Smith, B.J., and Bellingham, M.A., (2009). Integrated catchment management effects on flow, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill-country streams, New Zealand Journal of Marine and Freshwater Research, 43:3, 775-802, DOI: 10.1080/00288330909510041

Quinn, J.M. and Phillips, C. (2016). Production Forestry. In Advances in New Zealand Freshwater Science, P.G. Jellyman, Y.J.A. Dabie, C.P. Pearson and J.S. Harding (Eds), 469–481. Christchurch, NZ: New Zealand Hydrological Society and New Zealand Limnological Society.

Schmidt, J., Kienzle, S., and Srinivasan, M. (2009). Estimating increased evapotranspiration losses caused by irrigated agriculture as part of the water balance of the Orari catchment, Canterbury, New Zealand. *Journal of Hydrology (New Zealand), 48*(2), 73-94. Retrieved February 20, 2020, from www.jstor.org/stable/43944979

Shearer, K.A., and Young, R.G., (2011). Influences of geology and land use on macroinvertebrate communities across the Motueka River catchment, New Zealand, New Zealand Journal of Marine and Freshwater Research, 45:3, 437-454, DOI: 10.1080/00288330.2011.587823.

Statistics New Zealand. (2018). New Zealand Landcover. Published April 2019. www.stats.govt.nz.

Te Uru Rakau. (2019). One Billion Trees Fund: Six month monitoring and evaluation report. A monitoring and evaluation report on the state of the One Billion Trees Fund to 30 June 2019. New Zealand Government.

Thompson, R., Phillips, N., and Townsend, C.R. (2009). Biological Consequences of Clear-Cut Logging Around Streams – Moderating Effects of Management. Forest

Chapter 1.

The interactive effects of geology and land use on East Coast macroinvertebrate stream communities.

Abstract

Land use and geology both potentially influence streams and rivers by altering hydrology, water chemistry, light levels, sedimentation, channel form, food sources and habitat structure for the macroinvertebrate community assemblages. This study investigated the effect of geology and land use on stream macroinvertebrate communities in the East Coast region of the North Island, New Zealand. Macroinvertebrate communities were monitored every year between 2016-2018 at 79 sites during the austral summer in; exotic forest, indigenous forest and pasture streams that either had a hard or soft geology. Both land use and geology independently influenced biotic indices (MCI and %EPT abundance) and macroinvertebrate communities, but there seemed to be limited interactive effects. Exotic forest and pastoral land use were dominated by Molluscs and Ephemeroptera. Pastoral land use supported the highest number of macroinvertebrate individuals and indigenous forest had the highest densities of sensitive species. Sixty five percent of sites had deposited sediment levels greater than 20% cover. Sedimentation reduced the abundance of sensitive taxa, decreased biotic indices and was elevated in exotic forest, pastoral land use and soft geology. Calcium and conductivity levels were the parameters most strongly linked with macroinvertebrate

community structure, with higher conductivity levels in exotic forest, pastoral land uses and soft geology. These results demonstrate the complex relationship between geology and land use. This is important for future regional decisions as the combination of land use in a certain geology type could have extrapolated negative effects on macroinvertebrate assemblages indirectly through changes in habitat and water chemistry.

Introduction

Anthropogenic changes to land use, riparian vegetation and stream catchments can influence water chemistry, stream volume, habitat structure, sedimentation volume, volume of organic matter and light levels (Sweeney *et al.,* 2004; Quinn *et al.,*1997). These changes in turn alter stream substrate, organic matter volume, periphyton abundance, nutrient enrichment, stream shading and temperature and therefore can influence the abundance and diversity of freshwater communities (Townsend *et al.,*1983; Ormerod *et al.,* 1993; Reed *et al.,* 1994; Townsend *et al.,* 1997).

Streams draining pasture often have reduced ecological health and impaired invertebrate communities compared to streams in indigenous forest because of the changes outlined above (Quinn, 2000; Parkyn *et al.,* 2006). Streams in plantation forestry may have improved stream health compared to pasture streams as erosion may be reduced and sediment inputs less (Harding *et al*., 2000; Quinn, 2005; Marden *et al*., 2014). Forest harvesting however, has been shown to have negative effects on macroinvertebrate communities with a change from sensitive taxa to those more tolerant of degraded conditions (Davies and Nelson 1994; Death *et al.,* 2003; Harding *et al.,* 2000). The intensification of land use including dairy farming, mob-stock grazing and the associated non-point source discharges result in increases in nutrients, sediment and pathogens again degrading freshwater systems (Gillingham and Thorrold, 2000; Monaghan *et al.,* 2009; McDowell *et al.,* 2011). The integrity and functioning of stream health and macroinvertebrate communities is closely linked with both the adjacent riparian vegetation and the overall catchment land use (Allan, 2004; Quinn, 2000; 2005; Death and Collier 2010).

These changes in land use on stream ecosystems do not occur in isolation to other catchment characteristics, such as topography and geology that can interact with changes in land use (Allan & Johnson, 1997; Shearer and Young, 2011). Geology influences freshwater systems by controlling geomorphological processes, hydrology, water chemistry (dissolved minerals through erosion processes) and sedimentation (Dow *et al.,* 2006). These directly impact habitat characteristics for aquatic organisms and potentially alter macroinvertebrate assemblages (Minshall, 1984; Neff & Jackson, 2011; Shearer & Young, 2011; Dudgeon, 2012). However, to date most of the research on land use impacts have not investigated the potential interaction with geology.

With the increasing degradation of water quality it is important that we better understand the link between land use intensification and geology and the consequent influence on freshwater assemblages. Nationally and regionally, this information can be used by authorities to make more informed decisions on how best to manage water quality. Land use such as agriculture may have negative effects on stream catchments; however the magnitude of these effects may differ substantially between different geologies. The effects of similar land use practices in more erodible geologies could result in greater levels of sedimentation and a larger impact on freshwater communities.

The geology on the East Coast of the North Island is dominated by soft sandstone and mudstones (Fig. 1) and has some of the most erodible land in the country (Marden *et al*., 2014). There have not been any studies of the interaction between land use and geology in this region of New Zealand to the best of our knowledge. In this study I investigate the effect of land use on macroinvertebrate communities in hard and soft geologies across the region.

Fig. 1. Spatial distribution of lithologic/tectonic terrains of the East Coast region, North Island, New Zealand (Fig. from Marden *et al.*,2014).

Study Sites

The Gisborne region, located on the East Coast of the North Island, New Zealand, is approximately 8355 km^2, includes over 12,000 km of river length and is dominated by Cretaceous and Tertiary soft mudstone and sandstone geologies (Fig. 1), with 90% of the regions area being hill country terrain (Marden *et al.,* 2014)**.**

The soft geologies in the region have been particularly prone to erosion and the extent of the erosion in the East Coast is greater than in any other part of New Zealand (Phillips and Marden, 2005). Extensive grazing of sheep and cattle following indigenous forest land clearance in the 19th and 20th centuries has resulted in severe erosion and high sediment yields in many locations with currently 26% of the region considered to be actively eroding and 57% considered to have erosion potential (Bayfield and Meister, 1998; Jessen *et al*., 1999). The factors that influence erosion include; land cover, underlying geology, soil type, drainage network and the frequency of storms. These factors all dictate the intensity, location and type of erosion (Reid and Page, 2002).

The Gisborne region extends north to Lottin Point on the East Cape and south to the Wharerata range (Fig. 1). The Raukumara range boundaries the western side of the region and runs from the western boundary at Matawai, to the northern section of the region. Land use in the region is composed of 56% agriculture, 24% exotic plantation forestry, 13% indigenous forest (including regenerating) and 2% horticulture (Stats NZ, 2012) with the dominant proportion of the indigenous land cover located in the western Raukumara ranges (Price and Fitzgerald 2017). There are currently only 5 dairy farms, so pastoral agriculture is dominated by sheep and beef hill country farming.

Monitoring sites were chosen by Gisborne District Council (GDC) using a random selection protocol that included land use, source of flow, geology, stream order and climate from the River Environment Classification (REC2) system (Snelder and Biggs, 2002) (Table 1, Appendix 1). The Landcover was determined for each site using Larned *et al* (2017) and land use classified as either pasture, exotic forestry or indigenous forest. For this study, 79 sites were included in the analysis due to the extra two sites in the full list being under represented in land use (Fig. 2 & Fig. 3).

Table 1. Number of sites in the Gisborne biomonitoring region in River Environment Classification (REC) categories, Geology and Land use classifications (Larned *et al.*, 2017).

Geology	**Number of sites**	**Land use**	**Number of sites**
Hard	22	Exotic Forest	14
Soft	57	Indigenous forest	9
		Pasture	56
Total	79	Total	79

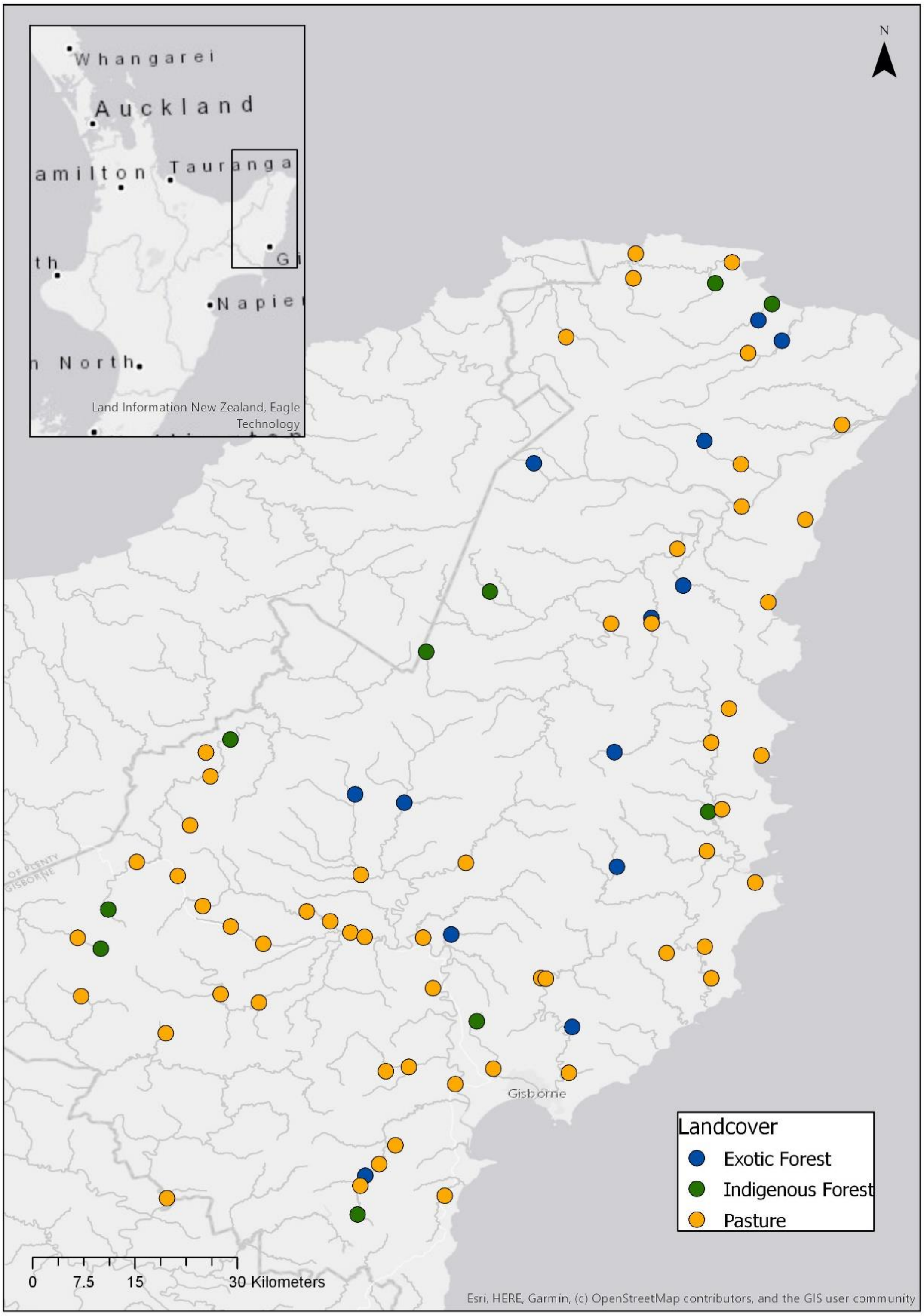

Fig. 2. Map of sampled sites with respective land use (indigenous forest, exotic forest, pasture) that were monitored for three years from 2016-2018 in the East Coast region, North Island, New Zealand.

Fig. 3. Photos of some sites monitored between 2016-2018 including: top left; plantation forestry, top right; pastoral, middle left; Pastoral, middle right; Indigenous forest, bottom left; plantation forestry, bottom right; pastoral

Methods and Materials

Invertebrate Sampling and Processing

Samples were collected from each site during the Austral summer between December and April during low flow conditions each year in 2016, 2017 and 2018. Macroinvertebrates were collected from all habitats within a 100 m stream reach using a kick net (30.5 by 20.4 cm net opening and 0.5 mm mesh net) with 5 kick samples of 0.2 m^2 collected over the reach and pooled. In the laboratory, samples were sieved, and 200 invertebrates enumerated and identified. A scan of the remaining sample for rare taxa (Duggan *et al.*, 2002) was then conducted and those invertebrates also identified. Macroinvertebrates were identified to the macroinvertebrate community index (MCI) level, generally genus.

Habitat and Periphyton Sampling

Periphyton assessments are conducted using a modification of the RAM-2 protocol (Biggs and Kilroy, 2000). Twenty visual assessments were conducted at each site using an underwater viewer and assessing the percent cover on the stream bed; the number of transects is dependent on stream width (<5m wide = 4 transects, 5-10m = 3 transects, 10-20m = 2 transects, >20m = 1 transect). Periphyton was characterised as; filamentous algae, thin film, mat, sludge, bare area, cyanobacteria.

Embeddedness was subjectively assessed as loose, good, moderate or tightly packed. Qualitative assessment of the habitat in the sample reach was performed with the Rapid Habitat Assessment Protocol (Clapcott, 2015). Habitat quality scores are based on percentage of deposited sediment cover, number of invertebrate habitat types, percentage of invertebrate habitat diversity, number of fish cover habitat types, percentage of fish cover, hydraulic heterogeneity types (e.g. pool, riffle, run), percentage of bank erosion, type of stream bank vegetation, riparian width and percentage of riparian shade. Each parameter was assessed with a score from 1 to 10 (with 10 being the best possible score). Scores are summed yielding a score out of 100 for each site. Water chemistry was measured with a YSI ProDSS multiparameter water quality metre (pH, Dissolved Oxygen %, Dissolved Oxygen mg/l, conductivity, salinity, turbidity, and temperature) recorded during sampling at each site.

Biological Indices

Biological metrics were calculated for each sample including the MCI (Boothroyd and Stark 2000; Stark and Maxted 2007) %EPT taxa (percent Ephemeroptera, Plecoptera and Trichoptera), and %EPT abundance. Both %EPT taxa and % EPT abundance exclude the caddisflies *Oxytheria* and *Paroxyethira* as they are relatively insensitive to pollution.

Data Analysis

Analysis of Variance was used in R (Version 3.5) to test whether land use or geology affected each of the biological metrics. Non-metric multidimensional scaling (NMDS) was performed on raw data using the Bray-Curtis distance measure in Primer version 7.0. Environmental variables collected at sites and from the Freshwater Ecosystems of New Zealand (FENZ) database (Leathwick *et al.*, 2010b) were correlated with the NMDS axes again using R.

Results

Macroinvertebrate Metrics

The geographical pattern of the MCI results reflects water quality higher in the catchments being excellent and having sensitive macroinvertebrate species present, to being degraded as you move down the catchments towards the sea (Fig.4).

% EPT abundance was highest at sites located in indigenous forest or exotic forest with 62% being the highest mean %EPT over the three years of sampling (Fig.5). Three sites had no EPT taxa present, eleven sites had EPT taxa greater than 50% and nineteen sites had EPT taxa less than 20%.

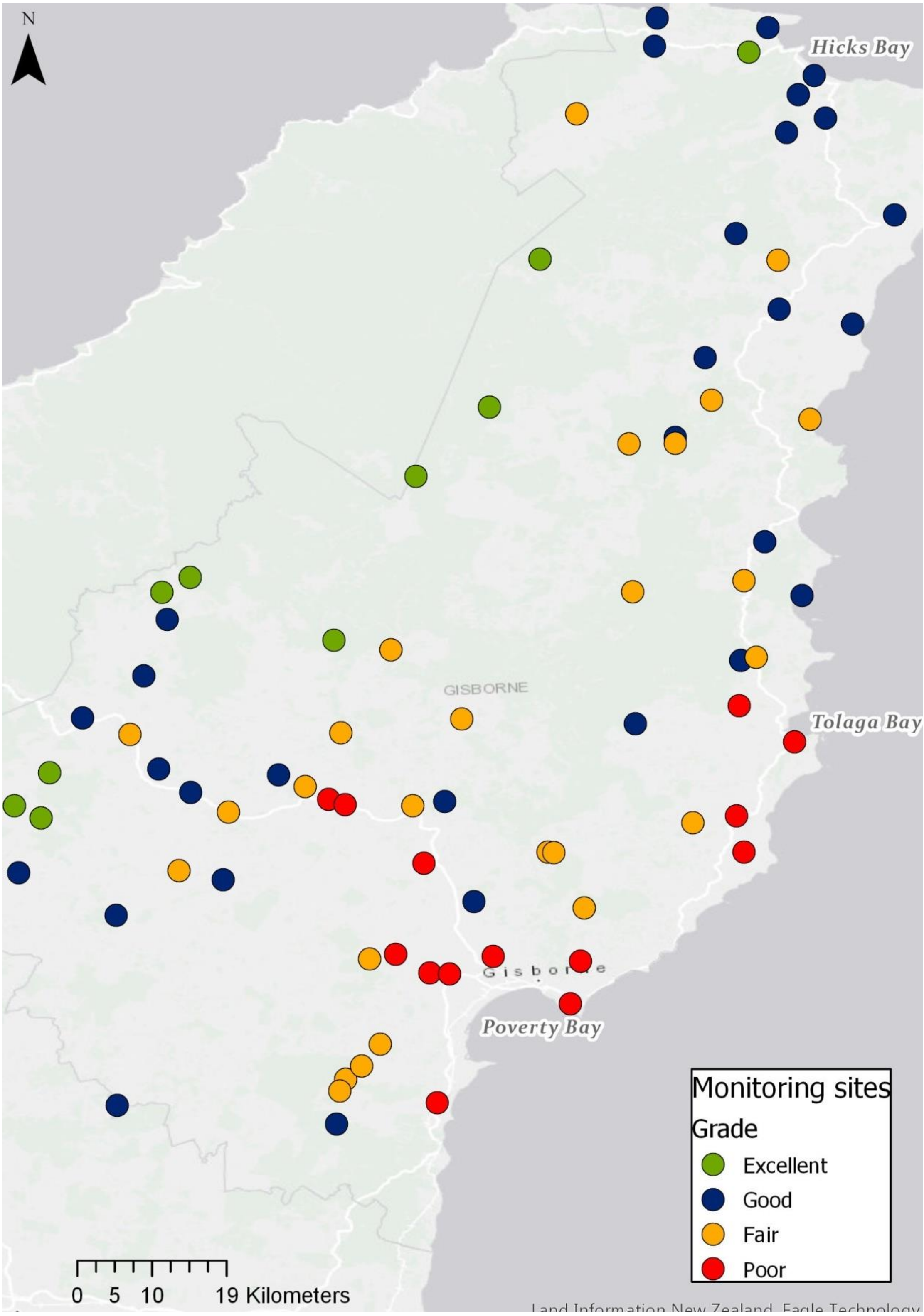

Fig.4. Mean MCI scores at 79 sampled sites in the East Coast region annually from 2016-2018, showing sites with excellent, good, fair and poor water quality classifications.

The Link Between Land use, Geology and Macroinvertebrate Metrics

%EPT taxa and %EPT abundance were higher in indigenous forest streams than in exotic forest or pasture streams by eight and twenty units respectively (Fig.5, Table 2). The MCI was also on average ten units higher in indigenous forest streams than in exotic forest or pasture streams (Fig.6, Table 2). There was no difference between the three sampling years. All three metrics (MCI, %EPT taxa and %EPT abundance) are influenced by geology with streams with hard sediment having higher metric scores (Fig. 7, Fig.8, Table 2). Land use and geology both have significant effects on all three metrics (MCI, %EPT taxa, %EPT richness) but there is no interaction between land use and geology (Table 2). The results reflect that both land use and geology influence the macroinvertebrate communities individually but there is no interaction between the two.

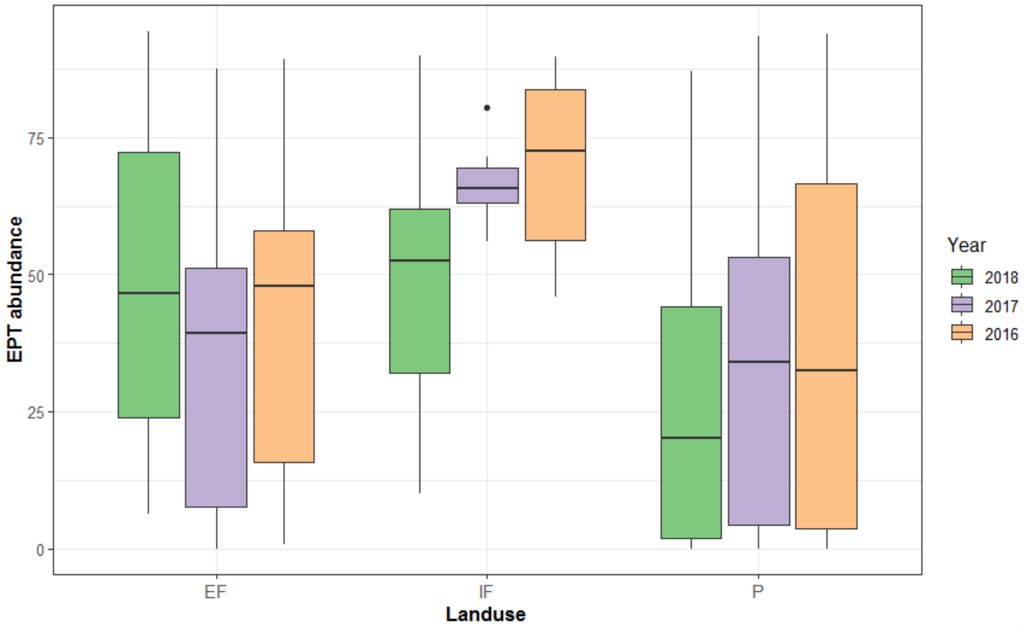

Fig.5. Boxplot of %EPT abundance in exotic forest, indigenous forest and pasture streams (EF=exotic forest, IF=indigenous forest, P=pasture) in the Gisborne region collected between 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%.

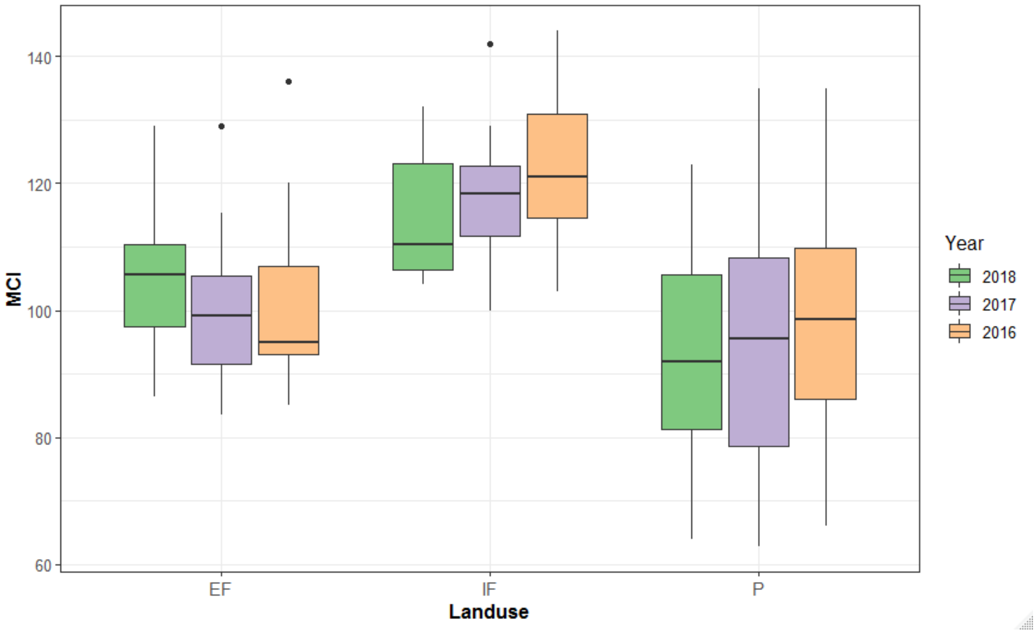

Fig.6. Boxplot of MCI in exotic forest, indigenous forest and pasture streams (EF= exotic forest, IF=indigenous forest, P=pasture) in the Gisborne region collected between 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%.

Table 2. Analysis of Variance results for macroinvertebrate metrics (MCI, %EPT taxa, %EPT abundance) from 2016-2018 testing for differences in indigenous forest, exotic forest and pastural land use and hard or soft benthic substrates for stream geology. Significant values are in bold.

			MCI		%EPT taxa		%EPT abundance	
	df	df	F	P	F	P	F	P
Land use	2	208	**35.82**	**<0.001**	**22.74**	**<0.001**	**35.82**	**<0.001**
Geology	1	208	28.83	**<0.001**	60.98	**<0.001**	28.83	**<0.001**
Year	2	208	2.3	0.13	1.69	0.19	3.11	0.07
Land use*Geology	1	208	2.3	0.098	3.31	0.07	2.81	0.06
Land use*Year	4	208	1.3	0.27	0.35	0.85	1.40	0.25
Geology*Year	2	208	0.012	0.91	0.64	0.53	0.03	0.87
Land use*Geology*Year	4	208	0.43	0.64	0.17	0.85	0.24	0.78

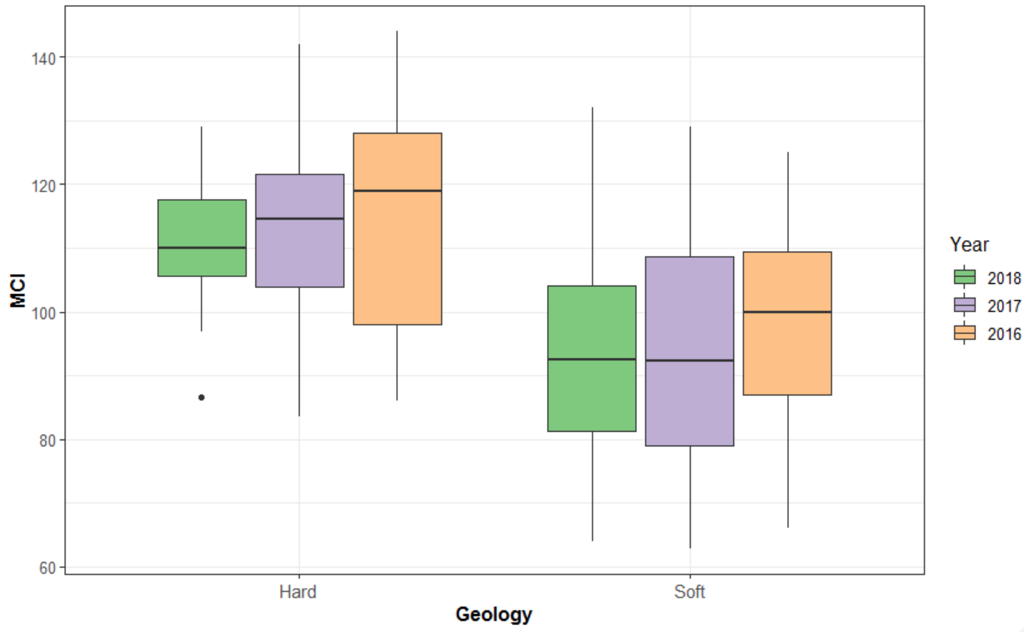

Fig. 7. Boxplot of %EPT abundance from streams in two geology types (Hard and Soft) collected in the Gisborne region from 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%.

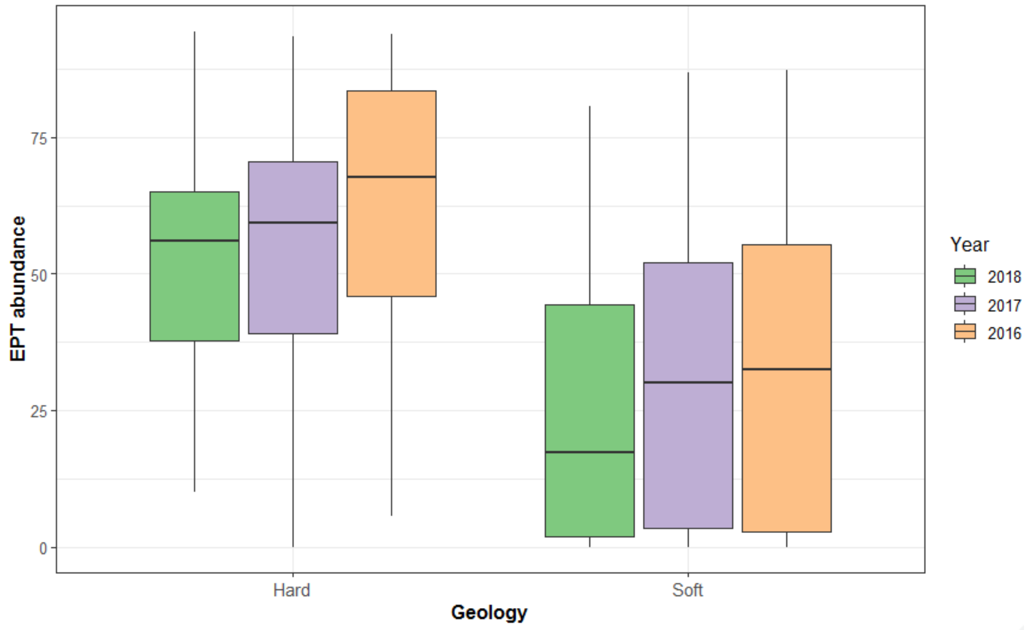

Fig.8. Boxplot of MCI values from streams in two geology types (Hard and Soft) collected in the Gisborne region from 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and

Geology and deposited sediment had a significant relationship ($F_{1,202}$= 31.15, P<0.001), with softer geology having a higher mean deposited sediment cover over the three years (Fig. 9). Exotic forest and pasture have the highest mean deposited sediment values (44% and 42% respectively) and indigenous forest has the lowest mean levels of deposited sediment (Fig. 10). Land use influenced deposited sediment cover ($F_{1,202}$= 8.08, P<0.001). MCI ($F_{1,208}$=36.09, P<0.001) and %EPT ($F_{1,208}$=43.04, P<0.001) declines as deposited sediment increases, (Fig.11).

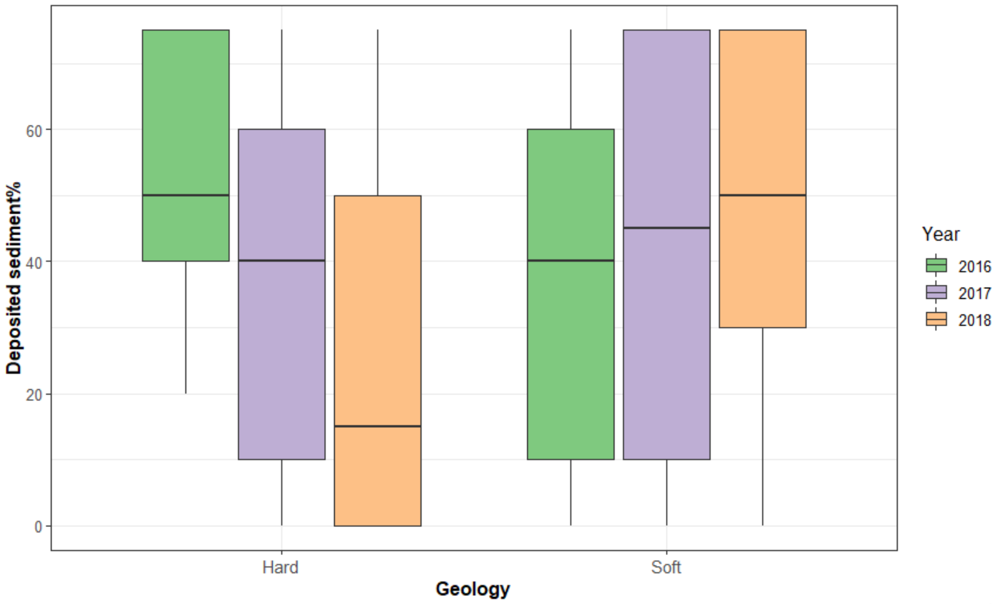

Fig. 9. Boxplot of percent deposited sediment from streams in two geology types (Hard and Soft) collected in the Gisborne region from 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%

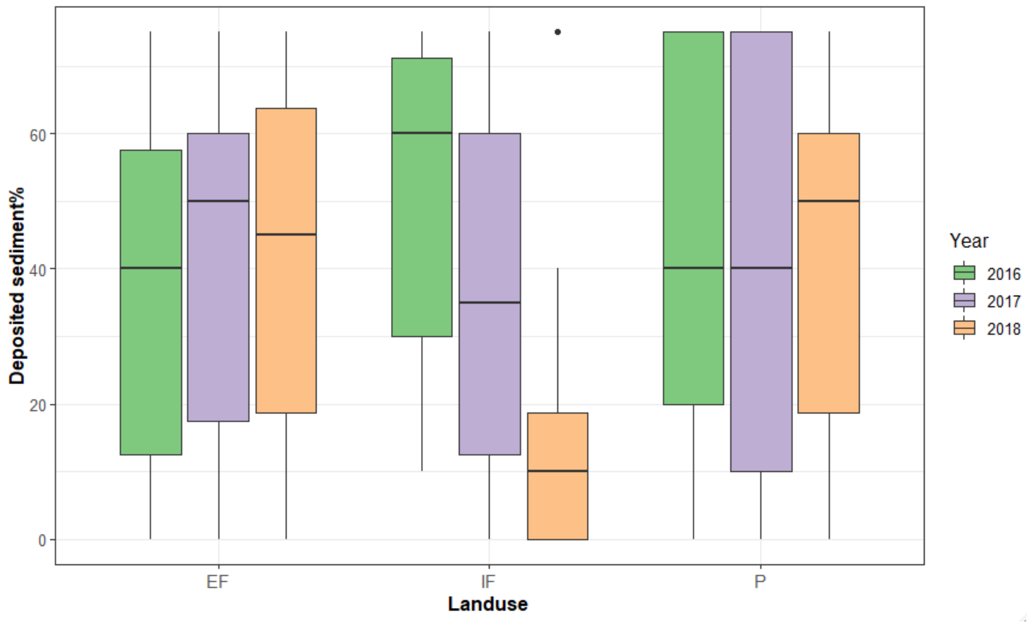

Fig. 10. Boxplot of deposited sediment in exotic forest, indigenous forest and pasture streams (EF= exotic forest, IF=indigenous forest, P=pasture) in the Gisborne region collected between 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%.

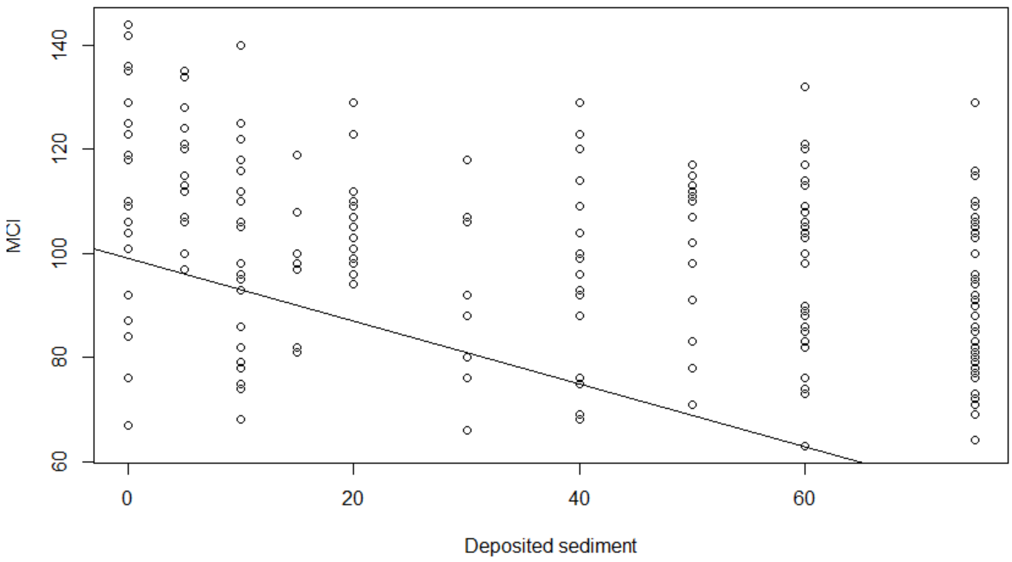

Fig.11. MCI plotted against percentage of deposited sediment for the Gisborne region collected from 2016-2018 ($F_{1,208}$=36.09, P<0.001).

Land use influenced conductivity ($F_{1,203}$=7.99 P<0.001). Conductivity levels in pasture streams ranged from 8µs/cm^{-1} to 1562µs/cm^{-1}, compared to those in indigenous forest which ranged from 64.9 µs/cm^{-1} to 625 µs/cm-1. Exotic forest streams had the highest median conductivity (561 µs/cm) (Fig.12). As conductivity increases the MCI declines ($F_{1,206}$=87.39 P<0.001) (Fig.13). Geology influences conductivity ($F_{1,206}$=31.36 P<0.001) with an increase in conductivity in soft geology streams (Fig. 14). Hard geology however had some of the highest conductivity readings (1562 µs/cm^{-1}); this was a site in pastoral land use. Soft geology conductivity readings range from (8.8 µs/cm^{-1} -1300 µs/cm^{-1}) with median values being 500 µs/cm^{-1}.

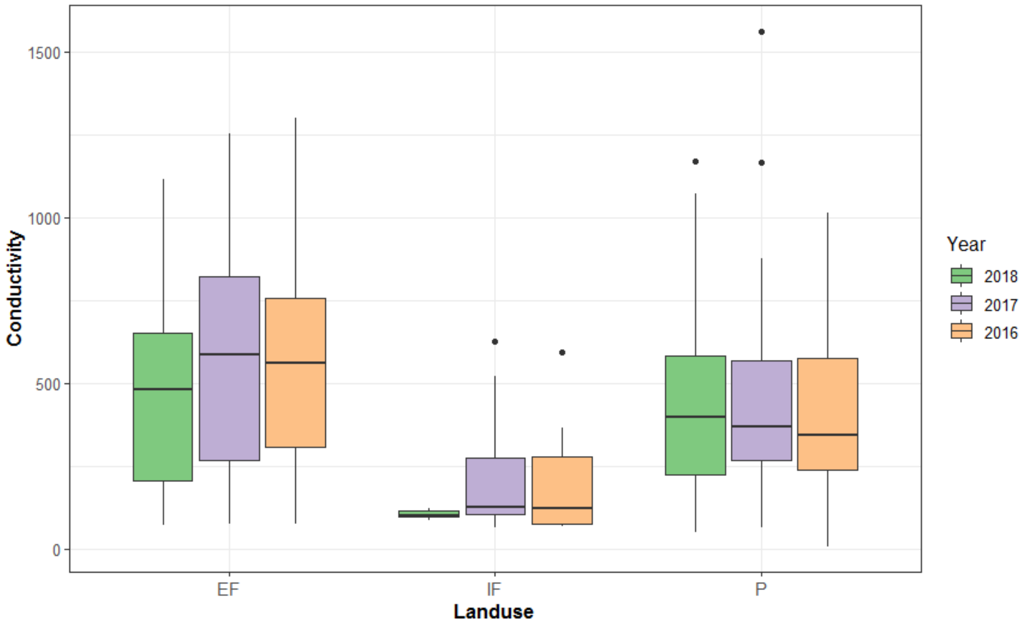

Fig.12. Boxplot of conductivity in exotic forest, indigenous forest and pasture streams (EF= exotic forest, IF=indigenous forest, P=pasture) in the Gisborne region collected between 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%.

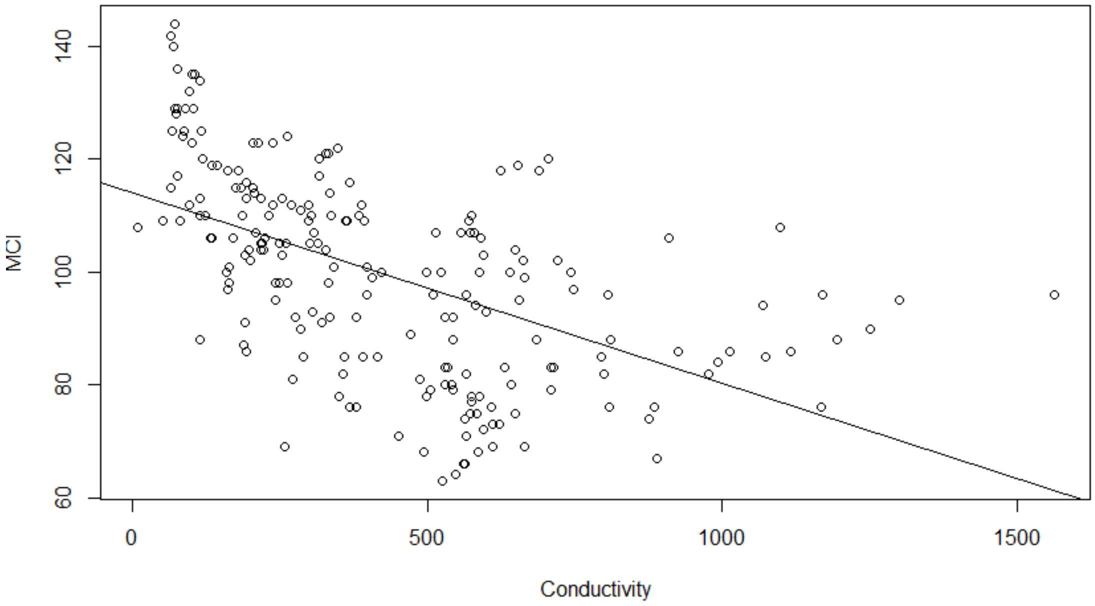

Fig.13. MCI plotted as a function of conductivity (µs/cm^{-1}) for samples collected in the Gisborne region between 2016-2018 ($F_{1,206}$=87.39 P<0.001, R^2=0.29).

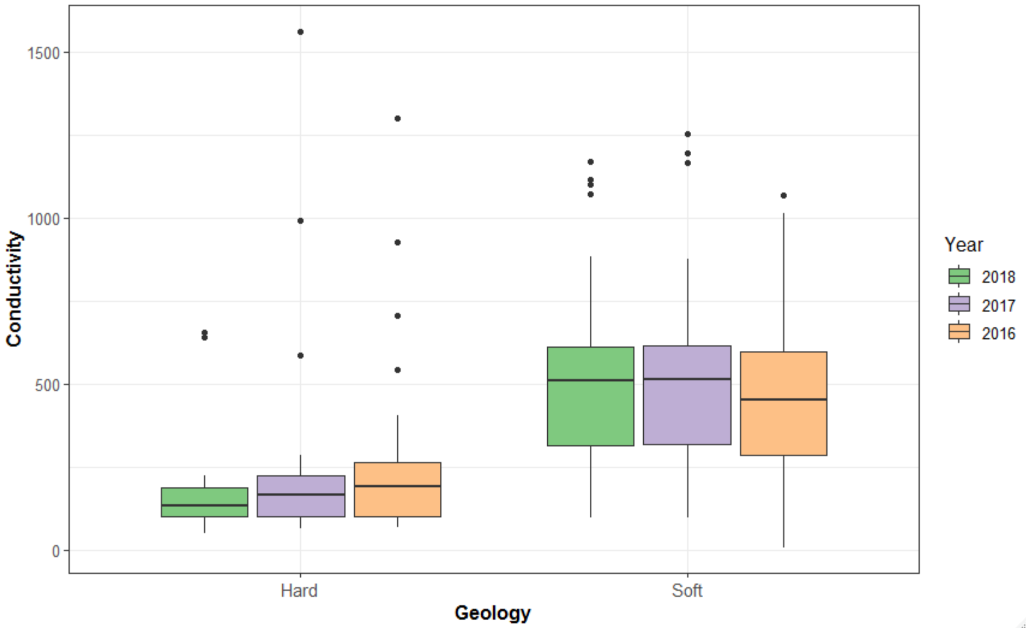

Fig. 14. Boxplot of conductivity values from streams in two geology types (Hard and Soft) collected in the Gisborne region from 2016-2018. Percentiles; boxes = 25% and 75%; Horizontal bars=5% and 90%; closed circles =5% and 95%).

Invertebrate communities

A total of 140 invertebrate taxa were collected at the 79 sites over the three years of sampling. The fauna was numerically dominated by the common freshwater snail *Potamopyrgus antipodarum*, mayfly *Deleatidium*, chironomid larvae *Tanyarsus* and Tanyponidae, Oligochaeta and caddisfly *Aoteapsyche*. Nineteen taxa were found to have relative abundances greater than 1%, but no one taxon was found at all sites in all three years.

The NMDS ordination had a low stress level of 0.19, indicating a strong difference in invertebrate community composition across the region, even though there was a core set of taxa that dominated samples (Fig.15). Pasture sites are predominantly clustered with similar community composition, with some sites having communities similar to exotic or indigenous forest. Indigenous forest communities were scattered with no sites being close to the dominant pasture cluster. Exotic forest communities ranged from some sites having similar community composition to pasture sites, and others different to both pasture and indigenous forest.

The two environmental variables that had the strongest positive correlation to macroinvertebrate community composition were upstream calcium (r_s=0.65) and conductivity (r_s=0.57) (Fig.15). The strongest negatively correlated variables were invertebrate habitat abundance ($r_{s=}$ -0.64), Upstream rain ($r_{s=-}$0.58), and deposited sediment ($r_{s=}$ -0.55).

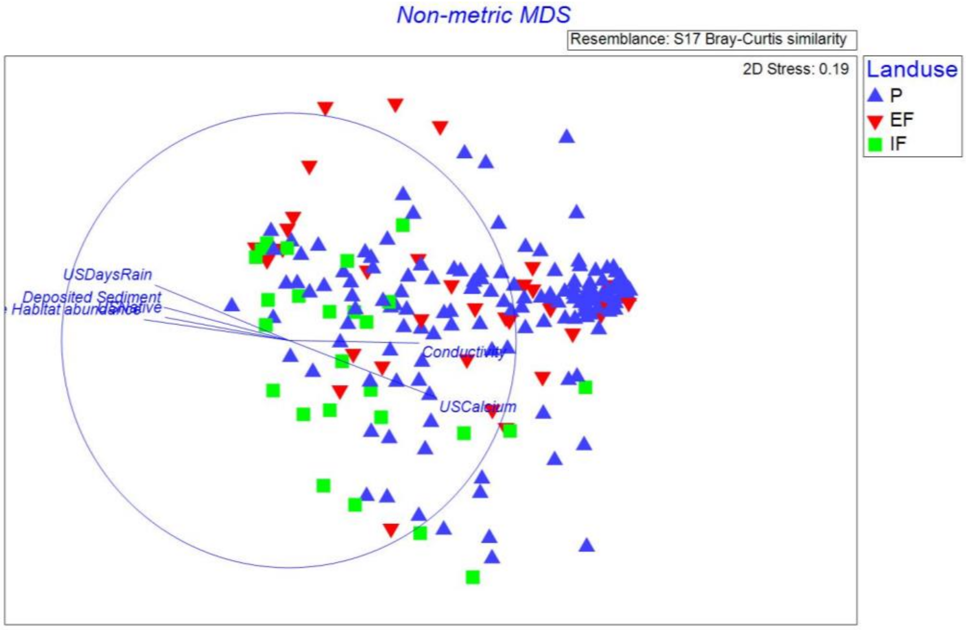

Fig.15. NMDS ordination of macroinvertebrate community composition at 79 sites from samples from 2016-2018. Environmental variables with a Spearman Rank correlation coefficient greater than $r_{s=}0.55$ are shown by the vectors overlaid.

Invertebrate densities were 5-fold greater in pasture compared to exotic and indigenous forest, because of the high numbers of Mollusca, Anthropoda and Ephemeroptera found in the pastoral streams (Fig. 16). Pasture had the highest number of invertebrates in all taxa apart from Plecoptera, where exotic forest had the highest densities. Indigenous forest was dominated by Ephemeroptera and had the lowest densities of Oligochaeta and Anthropoda. The number of sensitive species was highest in indigenous exotic forest.

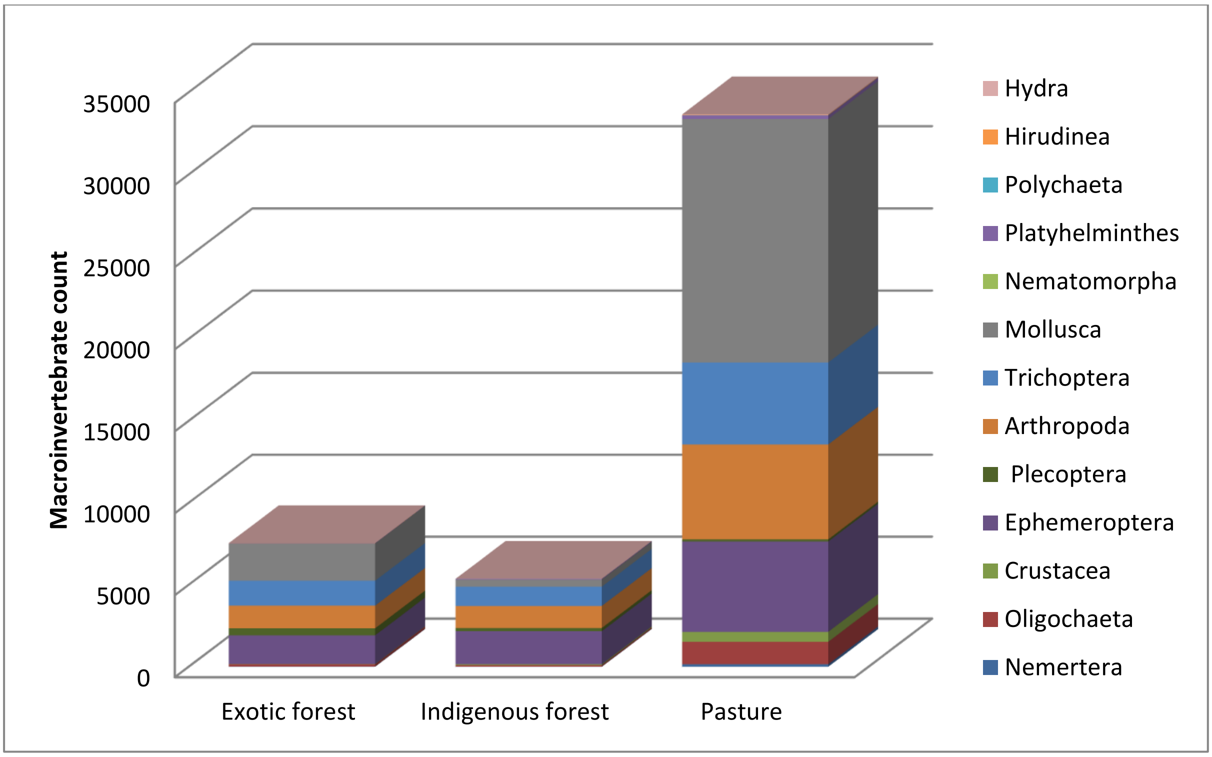

Fig. 16. Macroinvertebrate densities for each taxa in the three land uses collected in the Gisborne region from 2016-2018.

Discussion

This study investigated the impact of land use and geology on macroinvertebrate communities on the East Coast of the North Island, New Zealand. Macroinvertebrate communities had a range of species richness with indigenous and exotic forest land use supporting the most diverse and sensitive species assemblages of macroinvertebrates compared to pastoral streams. As the MCI is a metric that is based on organic enrichment there could be other environmental drivers influencing the MCI score such as geology and sedimentation.

Changes in macroinvertebrate communities have been linked to catchment land use and the nature of the riparian vegetation, in particular increased sedimentation, changes in substrate type and nutrients can all alter in-stream primary productivity rates (Quinn & Hickey, 1990a; Sweeney *et al.*, 2004). Land use clearly had an impact on these streams macroinvertebrate communities. Previous studies have shown that if land use changes to a more open system with less riparian vegetation and low inputs of nutrients, there will be little change to instream habitat and macroinvertebrate communities can become more diverse through an increase in primary productivity (Death and Zimmermann, 2005). Pastoral sites in this study not only have the lowest diversity but also some of the highest densities of sensitive species. The MCI reflected some pasture sites had excellent stream health that supported similar diversities and densities of sensitive species to the indigenous forested streams. The sites where this occurred were in extensive agriculture areas with a mixture of fragmented riparian vegetation.

Land use clearly has marked influences on streams in the Gisborne region. Although pasture sites were dominated by higher numbers of macroinvertebrate individuals, pasture was also dominated by an enrichment and sediment tolerant group of taxa (snails, chironomids, oligchaeates) and the numbers of sensitive taxa such as stoneflies were notably lower compared to indigenous and exotic forest land use. Chironomid abundance increases with periphyton biomass suggesting that the reduction of shade and therefore the increase in periphyton can be the main cause of the densities of chironomids in pastoral land use (Quinn and Hickey 1990a,b). The change in temperatures due to the reduction of shade also influences other invertebrate groups including Plecoptera, which could explain why they are less abundant in pasture land use. The communities of macroinvertebrates changed from being dominated by sensitive taxa in indigenous and exotic forest sites to pollution tolerant taxa in the pastoral and more intensively farmed areas of the region. Particularly in areas where cumulative downstream effects become concentrated.

In some areas in New Zealand indigenous forest and exotic forest can support similar macroinvertebrate communities (Townsend *et al.,* 1997; Quinn *et al.,* 1997). In other areas the underlying geology type has influenced a change in communities between indigenous and exotic forest, with exotic and indigenous forest having similar macroinvertebrate communities in granite, but not gravel geology (Shearer and Young, 2011). In this study, there was a difference in macroinvertebrate community composition between exotic and indigenous forest. Exotic forest streams were dominated by Molluscs (*Potamopyrgus*), Ephemeroptera (*Deleatidium*), and chironomids (*tanytarsini*) and indigenous forest was dominated by Ephemeroptera (*Coloburiscus, Tepakia*) and Dytiscidae.

Deposited sediment decreases instream habitat by filling interstitial spaces and increasing nutrient and bacteria levels. Deposited sediment cover of >20% negatively affects macroinvertebrate community assemblages (Burdon *et al.*, 2013). Increased fine sediment (<2 mm) levels decrease species richness and abundance with sensitive EPT taxa most negatively affected (Biggs, 2000; Ballantine and Davies-Colley, 2014; Ramezani *et al.*, 2014). In erosion prone geology on the East Coast of New Zealand, a decrease in vegetated area increases sedimentation into streams (Marden *et al.*, 2014). Therefore it is not surprising that streams with high deposited sediment levels negatively influenced the macroinvertebrate species by decreasing diversity and quantities of sensitive taxa and thus community composition. Over half (65%) of all samples taken had deposited sediment levels of >20% cover.

The two strongest driving variables that distinguished macroinvertebrate community assemblages were calcium and conductivity. These variables are both connected to geology and land use (Parkyn *et al.*, 2006). Weathering is higher in the soft (sandstone mudstone) geology, compared to the harder geology types (greywacke) and higher in land use that has reduced vegetated cover. The differences in weathering and erosion process result in a different chemical composition of the water (e.g. higher conductivity, increased pH, increased P), and in stream habitat (increased turbidity, increased suspended and deposited sediment), which indirectly influences macroinvertebrate communities through changes in primary productivity and habitat availability (Death and Joy, 2004; Neff and Jackson, 2011). The NMDS ordination showed the composition of communities at different sites was strongly influenced by the amount of calcium in the water column. Invertebrates use calcium for physiological and structural processes so the availability will determine the abundance of some organisms (Webster and Patten, 1979; Thorp and Covich, 1991). In turn that availability is dependent on the rock type (Webster and Patten,1979). This suggests that some rock types may be a limiting factor to certain species due to the amount of calcium that is available, gastropods for example are restricted to waters that have >25mg/l Ca (Thorp and Covich, 1991). A similar study comparing macroinvertebrate communities in different geological types also showed that the calcium and conductivity levels influenced macroinvertebrate community composition, with streams having low conductivity and calcium levels supporting different communities compared to those with higher levels (Neff and Jackson, 2011). The exotic forest land use had the highest levels of conductivity. Exotic forest on the East Coast is predominantly planted on the most erosion prone land to control land movement, the increased erosion rates and harvest cycle generating sedimentation of the waterways could be increasing mineral content and therefore conductivity and calcium levels which are influencing macroinvertebrate communities (Growns and Davis, 1991; Davies and Nelson, 1994; Death *et al.*, 2003). Indigenous forest, providing a stable vegetated riparian area and reduced erosion had the lowest conductivity levels with little variance compared to the other land uses.

Both land use and geology influenced the macroinvertebrate communities; however there did not appear to be any interaction. The impact of land use does not change between the geology types in the region. They both influenced macroinvertebrate communities in isolation which reflects that both are driving the changes in communities. This is important as the impacts of land use change are well known, such as the effects of changing from forest to pasture (Quinn and Hickey, 1990a; Death *et al*., 2003); however the influence of geology on macroinvertebrate communities irrespective of the land use is unclear. Other studies where the separation of geological and land use influences have attempted to disentangle their influences have also found it problematic (Kratzer *et al.,* 2006; Neff and Jackson, 2011). Geology often influences land use; in the Gisborne region land that was not appropriate for pasture has been used for exotic forest to reduce erosion (Marden *et al*., 2014). The locations of the exotic forest can therefore extrapolate the sedimentation of waterways, with stabilisation of the land during the growing phase and land disturbance during the harvest cycle. The geology also influences the intensiveness of the land with the more fertile geologies closer to the coast being used for more intensive land uses.

The investigation of the impact of land use and geology on the macroinvertebrate communities of the Gisborne region have shown how difficult it is to disentangle the interactions that land use and geology have on the macroinvertebrate communities. It is important to take into consideration both land use and geology type when managing the impacts of land use change or intensification and river health. Both have significant effects on macroinvertebrate communities through the physicochemical parameters and habitat of the streams, and therefore influence the communities in different ways. The findings of this study highlight that even in indigenous forested locations, geology is impacting habitat and can decrease sensitive EPT abundance. Further interrogation of specific habitat and water quality parameters, and splitting geology types into additional categories, may assist towards better understanding of what factors are determining specific interactions. Regardless, it is understood that land use and geology independently influence the community composition of macroinvertebrates through deposited sediment, conductivity, calcium levels and habitat abundance in the Gisborne region.

References

Allan, J.D. and Johnson, L.B. (1997). Catchment scale analysis of aquatic ecosystems. Freshwater Biology, 37:107-111.

Allan, J. D. (2004). Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics, 35, 257–284.

Ballantine, D. J., and Davies-Colley, R. J. 2014. Water quality trends in New Zealand rivers: 1989– 2009. Environmental Monitoring and Assessment 186:1939-1950.

Bayfield, M. A., and Meister, A. D. (1998) East Coast Forestry Project review: report for the Ministry of Agriculture and Forestry. Wellington, MAF.

Davies, P. E. and Nelson, M. (1994). Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research *45:*1289-1305.

Death, R., Baillie, B., and Fransen, P. (2003). Effect of Pinus radiata logging on stream invertebrate communities in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research. 37. 507-520. 10.1080/00288330.2003.9517185.

Death, R.G. and Zimmermann, E.M. (2005). Interaction between disturbance and primary productivity in determining stream invertebrate diversity. Oikos, 111, 392–402.

Death, R.G. and Joy, M. (2014) Invertebrate community structure in streams of the Manawatu–Wanganui region, New Zealand: the roles of catchment versus reach scale influences. Freshwater Biology 982-997.

Dow, C. L., Arscott D. B., and Newbold J. D. (2006). Relating major ions and nutrients to watershed conditions across a mixed-use, water-supply watershed. Journal of the North American Benthological Society 25:887–911.

Dudgeon, D. (2012). Responses of benthic macroinvertebrate communities to altitude and geology in tributaries of the Sepik River (Papua New Guinea): the influence of taxonomic resolution on the detection of environmental gradients. Freshwater Biology, 57: 1794-1812.

Gillingham, A.G., Thorrold, B.S., (2000). A review of New Zealand research measuring phosphorus in runoff from pasture. J. Environ. Qual. 29, 88–96.

Growns, I. O., Davis, J. A. (1991). Comparison of macroinvertebrate communities in streams in logged and undisturbed catchments 8 years after harvesting. Australian Journal of Marine and Freshwater Research 42*:* 689-706.

Harding, J. S., Quinn, J. M., and Hickey, C. W. (2000). Effects of mining and production forestry. *In:* Collier, K.C.; Winterbourn, M. J. *ed.* New Zealand stream invertebrates: ecology and implications for management. Christchurch, New Zealand Limnological Society. Pp. 230-259.

Jessen, M. R., Crippen, T., Page, M. J., Rijkse, W. C., Harmsworth, G. R., McLeod, M. (1999). Land Use Capability Classification of the Gisborne East Coast region. A report to accompany the second edition New Zealand Land Resource Inventory. Landcare Research Science Series No. 21. Lincoln, Manaaki Whenua Press.

Kratzer, E. B., Jackson, J. K., Arscott, D. B., Aufdenkampe, A. K., Dow, C. L., Kaplan, L. A., Newbold, J. D., and Sweeney, B. W. (2006). Macroinvertebrate distribution in relation to land use and water chemistry in New York City drinking-water-supply watersheds. Journal of the North American Benthological Society 25:954–976.

Larned, S., Snelder. T., and Unwin, M. (2017). Water Quality in New Zealand Rivers a modelled water quality state. National Institute of Water and Atmospheric research prepared for the Ministry for the Environment report CHC2016-070 MfE. (2001) A Guide to Riparian Assessment methods. Ministry for the Environment. Wellington. 34p.

Leathwick, J., West, D., Gerbeaux, P., Kelly, D., Robertson, H., Brown, D., Chadderton, W.L. and Ausseil, A.-G. (2010b). Freshwater Ecosystems of New Zealand (FENZ) Geodatabase. Users Guide. Department of Conservation, Wellington.

Marden, M., Arnold, G., Seymour, A., and Hambling, R. (2012). History and distribution of steepland gullies in response to land use change, East Coast Region, North Island, New Zealand. Geomorphology. s 153–154. 81–90. 10.1016/j.geomorph.2012.02.011.

Marden, M., Herzig, A., and Basher, L. (2014). Erosion process contribution to sediment yield before and after the establishment of exotic forest: Waipaoa catchment, New Zealand, Geomorphology, Volume 226, Pages 162-174.

McDowell, R. W., van der Weerden, T. J., and Campbell, J. (2011). Nutrient losses associated with irrigation, intensification and management of land use: a study of large scale irrigation in North Otago, New Zealand. *Agricultural Water Management, 98*(5), 877–885.

Minshall, G. W. (1984). Aquatic insect–substratum relationships. Pages 358-400 *in* V. H. Resh and D. M. Rosenberg (editors). The ecology of aquatic insects. Prager Publishers, New York.

Monaghan, R. M., Carey, P. L., Wilcock, R. J., Drewry, J. J., Houlbrooke, D. J., Quinn, J. M., *et al.*,. (2009). Linkages between land management activities and stream water quality in a border dyke-irrigated pastoral catchment. *Agriculture, Ecosystems & Environment*, *129*(1–3), 201–211.

Neff, M.R., and Jackson, D.A. (2011). Effects of broad-scale geological changes on patterns in macroinvertebrate assemblages, Journal of the North American Benthological Society 30, no. 2 (June 2011): 459-473.

Ormerod, S.J., Rundle, S.D., Lloyd, E.C. & Douglas, A.A. (1993). The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. *Journal of Applied Ecology*, 30, 13–24.

Parkyn, S.M.; Davies-Colley, R.; Scarsbrook, M.R.; Halliday, N.J.; Nagels, J.W.; Marden, M.; Rowan, D. (2006). Pine afforestation and stream health: a comparison of land-use in two soft rock catchments, East Cape, New Zealand. New Zealand Natural Sciences 31: 113-135.

Phillips, C.J., and Marden, M. (2005). Reforestation schemes to manage regional landslide risk.In: Glade, T., Anderson, M.G., Crozier, M.J. (Eds.), Landslide Hazard and Risk. JohnWiley and Sons Ltd., Chichester, UK, pp. 731–752.

Price R., Fitzgerald N. (2017). Gap Analysis and mapping of areas of biodiversity outside Protection Management Areas in Gisborne District. Landcare research Envirolink Grant: 1764-GSDC141.

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

Quinn, J. (2000). Effects of pastoral development. Pp208-229 in Collier K.J.; Winterbourn, M.J (eds) “New Zealand Stream Invertebrates; ecology and implications for management”, Caxton Press, Christchurch.

Quinn, J. (2005). Effects of rural land use (especially forestry) and riparian management on stream habitat. New Zealand Journal of Forestry. 49. 16-19.

Quinn J.M. and Hickey C.W. (1990b). Characterisation and classification of benthic invertebrate communities in 88 New Zealand rivers in relation to environmental factors. New Zealand Journal of Marine and Freshwater Research, 24, 387–409.

Reed, J.L., Campbell, I.C. & Bailey, P.C.E. (1994). Relationship between invertebrate assemblage available food at forest and pasture sites in three Iowa South-eastern Australian streams. *Freshwater Biology,* 32, 641–650.

Shearer, K.A., and Young, R.G., (2011). Influences of geology and land use on macroinvertebrate communities across the Motueka River catchment, New Zealand, New Zealand Journal of Marine and Freshwater Research, 45:3, 437-454, DOI: 10.1080/00288330.2011.587823.

Sweeney, B.W., Bott, T.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., Standley L.J., Hession, W.C. and Horwitz, R.J. (2004). Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences of the United States of America, 101, 14132–14137.

Thorp, J. H., and Covich A. P. (1991). Ecology and classification of North American freshwater invertebrates. Academic Press, San Diego, California.

Townsend, C., Arbuckle, C., Crowl, T., and Scarsbrook, Mike. (2003). The relationship between land use and physicochemistry, food resources and macroinvertebrate communities in tributaries of the Taieri River, New Zealand: A hierarchically scaled approach. Freshwater Biology. 37. 177 - 191. 10.1046/j.1365-2427.1997.00151.x.

Townsend, C.R., Hildrew, A.G. and Francis, J. (1983). Community structure in some southern English streams: the influence of physico-chemical factors. *Freshwater Biology*, 13, 521–544.

Webster, J. R., and Patten, B. C. (1979). Effects of watershed perturbation on stream potassium and calcium dynamics. Ecological Monigraphs 49:54-72.

Chapter 2.

The effect of forestry on stream macroinvertebrate communities in Gisborne, New Zealand

Abstract

The influence of forest harvest on macroinvertebrate communities in soft geology was investigated at fourteen stream sites in the upper Waipaoa River catchment, Gisborne, New Zealand. Stream reaches were surveyed under summer base flow conditions in March 2019 at sites that varied in time following forest harvest from recently harvested (0 years) to mature exotic forest (27 years). Macroinvertebrate communities were most influenced by turbidity, mean particle size, sediment size and bank erosion up to ten years following harvest. Macroinvertebrate communities following forest harvest were dominated by species sensitive to degraded stream conditions including Coleoptera and Diptera, with Diptera being one of the dominant taxa at all sites throughout the forest rotation. Fine sediment particle size did not change over the harvest cycle with all sites having silt present, and benthic sediment density decreasing as the forest established. The density of sensitive EPT taxa increased from ten years following forest harvest and communities became dominated by *Hydrobiosis*, *Deleatidium*, *Austroclima* and *Coloburiscus*. The QMCI increased with time since harvest with sites having high values after ten years. Recovery of macroinvertebrate communities was dependent on riparian shade, vegetation, hydraulic heterogeneity and habitat quality, with the recovery time being at least ten years following harvest. The results highlight the importance of retaining or establishing continuous riparian buffers along streams within harvested areas of forest to reduce negative impacts on freshwater communities.

Introduction

Pinus radiata forest plantations are managed as a rotational cycle that includes tree planting, growth of trees to a size appropriate for timber harvest (generally 28 years) and forest reestablishment. Plantation forestry has been used as a tool to manage erosion over large land areas throughout New Zealand and has been successful over the 28 year forest cycle in providing an effective solution to erosion control by re-stabilising and preventing movement and weathering of land (Marden *et al*., 2014). Exotic forests therefore have the potential to reduce erosion, sedimentation and further land disturbance (Glade, 1998; Reid and Page, 2003; Marden *et al*., 2014).

The most significant adverse effects on freshwater systems occur in the harvesting stage of the plantation forestry cycle (Harding *et al*., 2000, Death *et al*., 2003). Clear-fell harvest leaves areas of exposed sediment that are susceptible to erosion from rainfall and surface runoff (Ballie *et al*., 2005). Site characteristics can influence the size and duration of effects (e.g., geology, climate, soils) as well as harvest methods and practice techniques such as; road construction, ground and hauler harvest practice and riparian management (Harding *et al*., 2000; Parkyn *et al*., 2003; Ballie *et al*., 2005; Reid *et al*., 2010). The effects on freshwater systems from forestry practices include; increased sedimentation, increase in turbidity and alkalinity, altered water yield, increased nutrient inputs from particulate matter in surface runoff, change in channel morphology, reduced or loss of riparian margins and decreased organic matter in stream (Harding *et al*., 2000; Ballie *et al*., 2005; Parkyn *et al*., 2006; Reid *et al*., 2010). Increased sediment has the potential to decrease stream substrate size, reduce interstitial spaces and can cause habitat loss for freshwater species (Harding *et al*., 2000; Death *et al*., 2003; Ballie *et al*., 2005; Quinn and Wright-Stow 2008). Sediment can be retained on harvested slopes through a quick turnaround of replanting and ensuring ground cover, but it can take several years for the sediment to be flushed from the streams, riparian vegetation to establish, and for stream inputs to return to pre-harvest levels (Moore *et al*., 2005; Quinn and Wright-Stow, 2008).

Previous studies in New Zealand have indicated that stream size, harvest debris management and regrowth of riparian vegetation influence the time for stream recovery after clear-fell logging (Moore *et al*., 2005; Quinn and Wright-Stow, 2008). The time taken for streams to recover has varied from two to sixteen years; a third to half of the pine forest rotation time, for full stream temperature recovery which is dependent on stream size (Harding *et al*., 2000; Quinn and Wright-Stow, 2008; Reid *et al*., 2010; Death, 2017). The magnitude of effects on stream health is larger when more of the stream edge is harvested and when streams are fed by overland flow rather than springs (Ballie *et al*., 2005; Davies *et al*., 2005). Clear-fell, catchment scale harvest methods are a common harvest practice method in New Zealand, however progressive smaller scale logging may have less effects on stream communities (Davies *et al*.,2005; Reid *et al*., 2010).

Long term effects on streams in harvested areas can include changes to stream morphology, increased channel width, reduced shading and loss of in stream physical habitat (Bilby and Ward, 1991; Harding *et al*., 2000). The results of these physical changes in turn may reduce the prevalence of sensitive Ephemoptera, Plecoptera and Trichoptera species and increase that of pollution tolerant species; Mollusca, Oligochaeta, and Diptera (Harding *et al*., 2000; Davis, 2001; Death *et al*., 2003). The changes to macroinvertebrate communities following forest harvest have included an increase in abundance (Quinn and Halliday, 1999), no change between logged and unlogged sites (Noel *et al*., 1986), or an increase in sensitive species due to an increase in allochthonous inputs (Silsbee and Larson, 1983) and declines in sensitive species following harvest similar to communities found in pasture (Death *et al*., 2003). Previous studies on the impacts to macroinvertebrate communities following forestry harvest have almost all been conducted in regions with hard, less erodible geology such as greywacke (Death *et al*. 2003; Reid *et al*., 2010). Geomorphological context could be responsible for differing results of freshwater communities following harvest in different regions. The rate of sedimentation, canopy closure, stream power, channel morphology and sediment composition could all influence the response of macroinvertebrate communities to forestry harvest and subsequent recovery.

I am not aware of any studies to investigate the effects of forest harvest on streams in soft geology such as the Gisborne region. The soft geology and accelerated erosion could potentially increase the negative effects of sediment on these streams. It is unclear if the magnitude of impact on macroinvertebrate communities following forest harvest is greater in soft geology streams, or if the macroinvertebrate communities take longer to recover. The objective of this study was to investigate the effect of forest harvest on stream macroinvertebrate communities in the soft geology of the Gisborne region.

Study Sites

Sampling sites were located in the Waipaoa and Mangatu forests, an exotic forest plantation which is predominantly *Pinus radiata* located in the upper Waipaoa River catchment on the East Coast of the North Island, New Zealand (Fig. 1). The catchment is 200 – 800 m above sea level with a humid temperate climate but is subject to weather systems (La Niña) that can result in severe rainfall or heavy drought (El Niño). Annual average rainfall is ~800 mm to

~2500 mm. The catchment is steep with the steeper areas averaging more than 35° (Marden *et al.*, 2014).

The catchment geology comprises Cretaceous and Tertiary (24%) mudstone and sandstone (76%) (Marden *et al.*, 2014) which when exposed to weathering, results in dramatically reduced rock strength and high rates of erosion (Pearce *et al.*, 1981; Page *et al.*, 2001). The Waipaoa catchment yields 15 million tonnes of sediment per annum (Hicks *et al.*, 2000) which is high on global standards.

Fig. 1. The upper Waipaoa catchment showing major lithologies and gullies (after Marden *et al*., 2005), TG refers to the Tolaga Group (Fig. from Fuller and Marden 2011).

Indigenous land clearance from 1880-1920 in the catchment resulted in the loss of root strength and land stabilisation and caused accelerated erosion (O'Loughlin, 1974) and the forming of gully systems. The gully systems are one of the main sediment generating mechanisms along with; landslides, slumping, earthflow, sheet, tunnel and stream bank erosion (Page *et al*., 2001).

The increased rates of sediment delivery from the 1900s changed channel substrate from cobble dominated to gravel and sand substrate, along with channel widening and bed aggradation (Black, 1977; Hamilton and Kelman,1952). One of the most effective mechanisms to mitigate erosion and heal gully systems is reforestation. A reforestation programme was established in the 1960s to plant 140 km^2 of the upper Waipaoa catchment in exotic forest trees, predominantly *Pinus radiata* (69%), with some *Pseudotsuga menziesii* (8.5%), and *Acacia melanoxylon* (3.5%) (Marden *et al*., 2014). The planting of *Pinus radiata* on the steep slopes resulted in a fast growing species that accelerated evapotranspiration and lowered soil moisture, increased root cover and therefore a decreased rate of slope failure (Phillips *et al*., 1990; Marden *et al*., 2012). Gully systems have contributed significant sediment loads to the Waipaoa River from the 1960s c.17% of the annual sediment load to c 43% up to 1997 when reforestation was instigated (Marden *et al*., 2005; Marden *et al*., 2008).

Marden *et al*., (2014) showed reforestation was successful over the period from 1939-1988 at stabilising landslides and earthflows and decreasing erosion in the catchment. Land stabilisation became effective 8-10 years after planting where canopy closure occurred resulting in a reduction of eroding land area of ~62% with a ~51% reduction in the erosion rates in the upper Waipaoa catchment (Fig. 2). These reductions are estimated to be equivalent to a 12% reduction in sediment yield in the Waipaoa River. The major gullies however were not stabilised as a result of the upper catchment planting and continue to be the dominant source of sediment.

Fig. 2. Mapped distribution of erosion features including gullies, earthflows, shallow landslides and slumps interpreted as sites of active sediment generation in 1939, 1960 (pre-reforestation)1970, and 1988 (reforestation period). Striped area shows extent of reforestation plantings as at 1970 and 1988. Photography flown in 1988 captures slope failures initiated during CycloneBola in 1988.(Fig. from Marden *et al*.,2014)

The establishment of the 28 year rotation *Pinus radiata* forest in the upper Waipaoa catchment has resulted in a cycle of clear-fell harvesting and replanting trees and a mixture of first and second rotation trees. The 14 study sites are located in headwater 1st to 3rd order streams throughout the upper Waipaoa catchment, in areas that range from recently harvested through to mature forest, ready for harvest (Fig. 3, Table 3, Appendix 2), all sites are second rotation forest. The study sites are located upstream from known gully locations where possible, to avoid measuring gully sediment inputs. Sites had predominantly exotic forest upstream with some sites having small areas of indigenous vegetation.

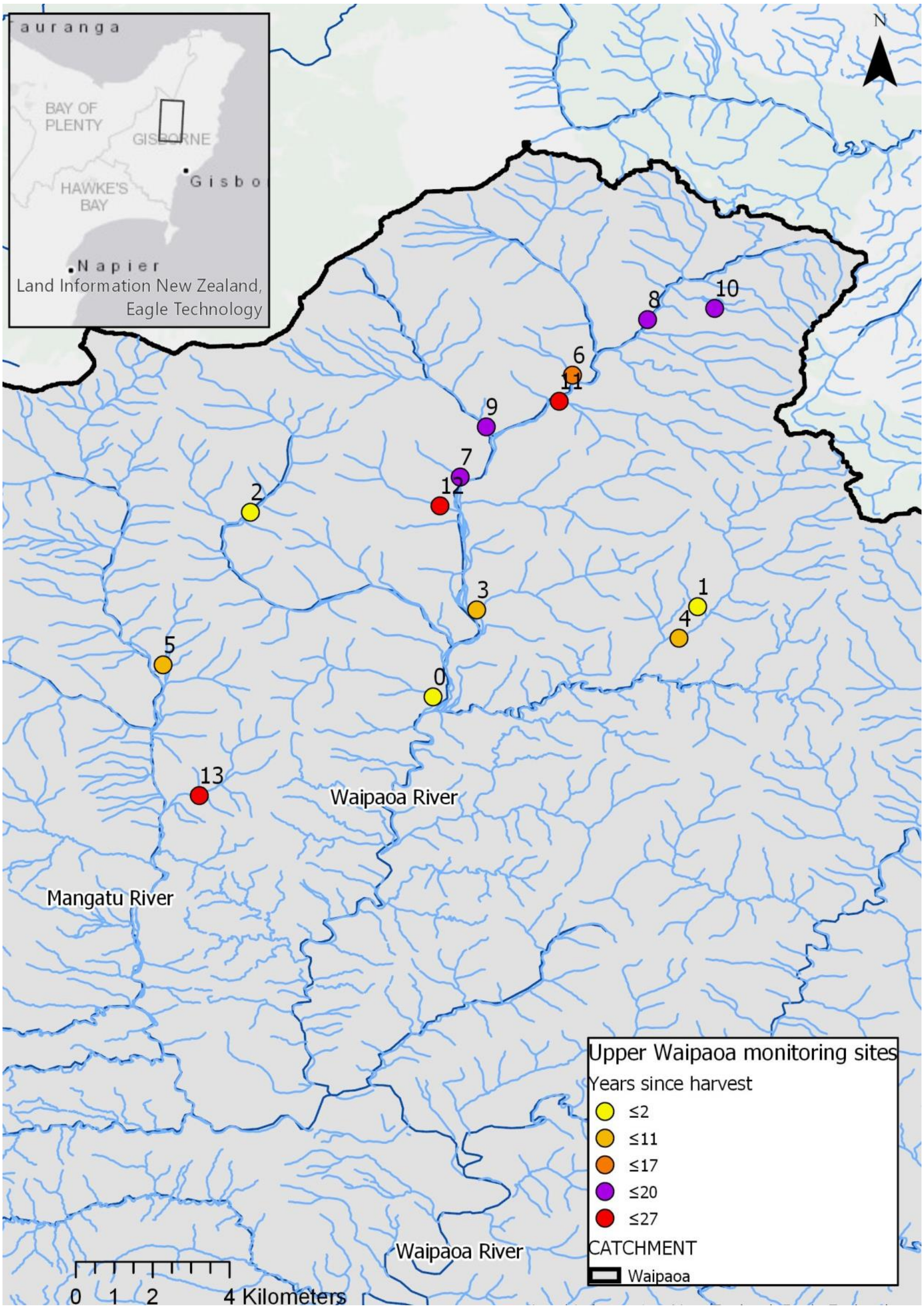

Fig. 3. Fourteen monitoring sites sampled in the Upper Waipaoa catchment in March 2019 showing years since forest harvest for each site.

Table 3. Site location, stream order, the year exotic forest was planted and number of years since forest harvest for fourteen sites sampled in the upper Waipaoa River catchment, Gisborne in March 2019.

Site number	Years since harvest	Established year	NZTM E	NZTM N	Stream Order	NZReach	Sample date
0	1	2017	2024417	5751966	3	5007060	6/03/2019
1	2	2016	2030095	5754077	1	5006646	7/03/2019
2	2	1991-2016	2020837	5757074	3	5006448	7/03/2019
3	3	2015	2025458	5754235	2	5006749	6/03/2019
4	3	2015	2029662	5753252	3	5006865	7/03/2019
5	11	2007	2018791	5753121	2	5006910	12/03/2019
6	17	2001	2027802	5760375	2	5006027	6/03/2019
7	18	2000	2025297	5757780	2	5006359	5/03/2019
8	18	2000	2029457	5761765	3	5005917	5/03/2019
9	18	2000	2025910	5759093	3	5006188	5/03/2019
10	20	1998	2030881	5761985	2	5005897	6/03/2019
11	22	1996	2027485	5759693	3	5006155	6/03/2019
12	23	1995	2024827	5757044	2	5006440	5/03/2019
13	27	1991	2019372	5749604	3	5007414	8/03/2019

Methods

Macroinvertebrate collection

Macroinvertebrates were sampled once during March 2019 at fourteen sites. At each site, five 0.1m^2 randomly placed Surber samples (250µm mesh) were collected from riffle and run habitat and preserved in 70% isopropyl alcohol (Stark *et al*., 2001). Samples were collected over a 50-100 m reach of stream. In the laboratory samples were sieved (500µm mesh) and invertebrates were identified and counted to the lowest possible taxonomic classification using available keys (Winterbourn 1973 ; McLellan 1999; Winterbourn *et al*., 2006). Chironomids were not identified below family.

Physicochemical and habitat characteristics

Streambed particle size was determined using the Wolman pebble count protocol (Clapcott *et al*., 2011) and classifying >100 particles according to the Wentworth scale size (bedrock to clay/silt). A ruler was used to measure stream depth at transect points across the stream width, and depths were averaged to one depth measurement.

Qualitative assessments of habitat were conducted at each site using the Rapid Habitat Assessment Protocol (Clapcott, 2015) for the 50-100 m reach sampled. Habitat quality scores are based on percentage of deposited sediment cover, number of invertebrate habitat types, percentage of invertebrate habitat diversity, number of fish cover habitat types, percentage of fish cover, hydraulic heterogeneity types (e.g. pool, riffle, run), percentage of bank erosion, type of stream bank vegetation, riparian width and percentage of riparian shade. Each parameter was assessed with a score from 1 to 10 (with 10 being the best possible score for each parameter). Scores are added together resulting in a score out of 100 for each site, with 100 being the highest possible score.

In-situ water quality measurements were made at the time of collection using a YSI ProDSS multiparameter water quality metre (pH, Dissolved Oxygen %, Dissolved Oxygen mg/l, conductivity, salinity, turbidity, and temperature) at each site.

Periphyton assessments are conducted using a modification of the RAM-2 protocol (Biggs and Kilroy 2000); 20 visual assessments are conducted at each site using a viewer and measuring percent cover of periphyton on the stream bed. The number of transects is dependent on stream width (<5m wide = 4 transects, 5-10m = 3 transects, 10-20m = 2 transects, >20m = 1 transect). Periphyton was characterised as; filamentous algae, thin film, mat, sludge, bare area, cyanobacteria.

Embeddedness was subjectively assessed as loose, good, moderate or tightly packed.

Wetted width was measured and the length of meso habitats (riffle, run, pool) were measured and recorded along the sample reach. The maximum depths of the pools in the sample reach were recorded, as was the depth of the pool crest (the point where the pool forms a riffle/run) and the depth of soft sediment was recorded in pools. The floodplain channel shape was drawn for each site, and a channel cross section was measured with depth measurements taken from the left bank full height at regular intervals across the transect to right bank full height. Water depth was recorded at 10 transect points within the wetted width of the stream.

Sediment sampling

The amount of fine deposited sediment was measured by assessing through a viewer at 20 transects in the stream. Deposited sediment cover was recorded from 1% to >50% cover. Deposited sediment cover was then averaged across all transects and one average deposited sediment cover for the sample reach was calculated.

Sediment samples were taken from one location at each site in the centre of the stream using a Quorer. This was inserted to the bed of the centre of the stream, the upper 5-10 cm of substrate was disturbed for 15 seconds and a sample of the dirty water was taken in a screw top jar. In the laboratory samples were analysed using the Horiba LA_950V2 particle size analyser with the slurry sampler attachment. For consistency and in light of time constraints, samples were not treated for organic material and were run in the instrument using the Fraunhofer method which excludes all clay grain size fractions (<2 micron). Two samples were not able to be analysed due to the presence of organic material in the sample (site 6 and 13).

Biological indices

Biological metrics were calculated for each sample including the QMCI (Boothroyd and Stark 2000; Stark and Maxted 2007) %EPT taxa (percent Ephemeroptera, Plecoptera and Trichoptera), and %EPT abundance. Both %EPT taxa and % EPT abundance exclude the caddisflies *Oxytheria* and *Paroxyethira* as they are relatively insensitive to pollution.

Statistical analysis

Sediment sampling analysis was completed using the GRADISTAT (Blott and Pye, 2001). This provides; sieving, mean, mode, sorting, kurtosis and skewness calculated arithmetically and geometrically (in metric units) and logarithmically (in phi units) using moment and Folk and Ward graphical methods. The mean grain size refers to the average grain size of the sample. Table 4 shows the sorting classifications for the samples and the Kurtosis distribution classifications, wide ranges of sediment sizes have poor sorting, and samples with a small grain size range are well sorted.

Table 4. Skewness classification of soil and Kurtosis classifications for the range of sediment sizes analysed by GRADISTAT

Sorting Range (φ)	Description of	Kurtosis Range	Description of
< 0.35	Very well sorted	< 0.67	Very platykurtic
0.35 – 0.50	Well sorted	0.67 – 0.90	Platykurtic
0.50 – 0.71	Moderately well	0.90 – 1.11	Mesokurtic
0.71 – 1.00	Moderately sorted	1.11 – 1.50	Leptokurtic
1.00 – 2.00	Poorly sorted	1.50 – 3.00	Very leptokurtic
2.00– 4.00	Very poorly sorted	> 3.00	Extremely
> 4.00	Extremely poorly		

The Kurtosis distribution is the peakedness of the grain size distribution curve. Leptokurtic indicates a better organisation of the central distribution of grain sizes relative to the tails.

Results

Physicochemical results

The fourteen streams were all small streams with width ranging from one to four and a half meters and stream depth from 10 to 30 cm. All sites have high conductivity (222-1034 µS/cm) and pH (mean 8.3). Deposited sediment ranged from 5-55% cover (Table 5).

Table 5. Physicochemical characteristics of 14 sites sampled in the upper Waipaoa catchment, Gisborne, sampled in March 2019 in varying years post-harvest. Temp=Temperature, Cond=Conductivity, DO= Dissolved oxygen.

Site number	Year since harvest	Total habitat score	SAM4 deposited sediment %	Temp °C	Cond µS/cm	DO mg/l	DO %	Turbidity	PH	Average Depth (m)	Average width (m)
Site 0	1	50	12.95	11.8	949	10.3	96	2.6	8.1	0.1	1
Site 1	2	47	30.25	19.5	322.6	8.62	94	14.6	8.5	0.2	2
Site 2	10	52	6.65	25.3	419.2	7.98	97.1	0.7	8.53	0.07	3.5
Site 3	3	28	54.5	27.7	1026	8.59	100.3	5	8.25	0.01	1
Site 4	3	53	20.45	15.6	297.1	9.44	96.6	8.8	8.56	0.3	4.5
Site 5	11	50	6.55	15.2	598	11.24	112.2	0.9	8.51	0.1	1
Site 6	17	74	21.5	14.1	451.9	9.85	98.9	0.6	8	0.05	1
Site 7	18	75	6.95	13.9	273.9	9.99	96.8	7.3	8.26	0.15	1
Site 8	18	64	10	21.3	222	8.76	99.2	2.9	8.61	0.1	4
Site 9	18	59	9.25	20.9	308.6	9.3	104.3	4.1	8.09	0.05	4
Site 10	20	70	49.6	13.9	388.6	9.69	93.7	4.6	8.43	0.15	2
Site 11	22	58	10.55	13	501	10.1	97	28.4	8.54	0.3	2.5
Site 12	23	67	4.25	13.7	643	10.25	98.9	1.5	8.14	0.08	1
Site 13	27	43	14.6	16.6	1034	8.76	90.2	0.7	8.06	0.1	1

Temperature, conductivity and deposited sediment were lower in the more established forests (Fig. 4 and Fig. 5, Table 6). Turbidity was unaffected by forest maturity (Table 6). All streams were similar in pH and dissolved oxygen across all sites. Habitat scores increased as the years since harvest increased (Table 6, Fig. 6).

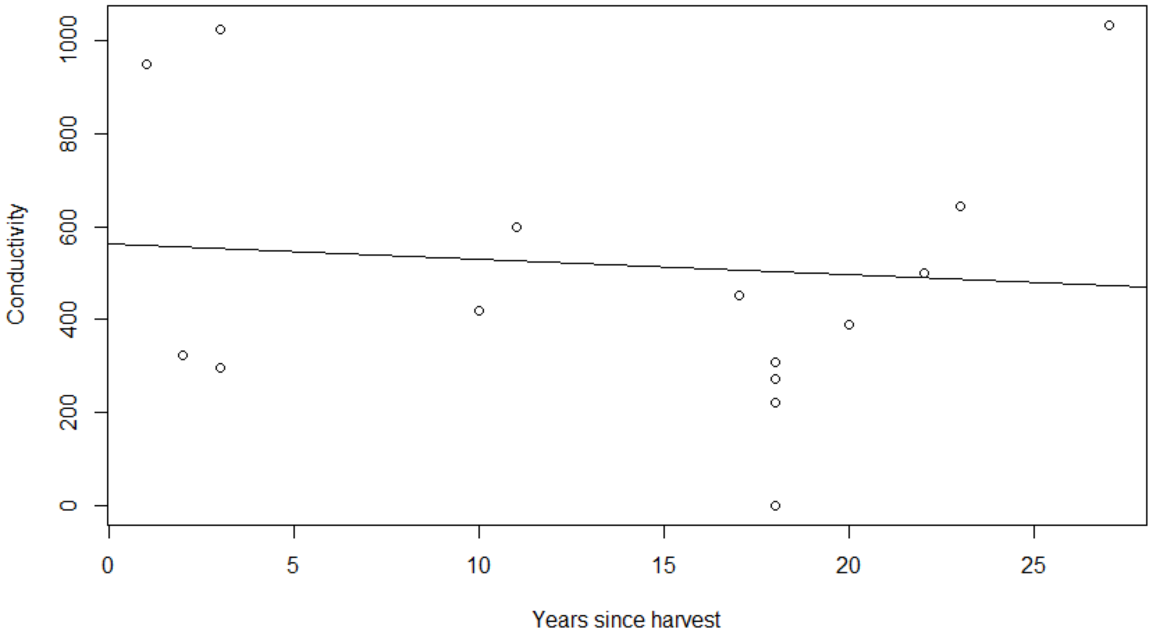

Fig. 4. Conductivity plotted against years since harvest for the 14 sites sampled in the upper Waipaoa catchment, Gisborne, New Zealand in March 2019 ($F_{1,68}$=0.67 P=0.04, R^2=0.009).

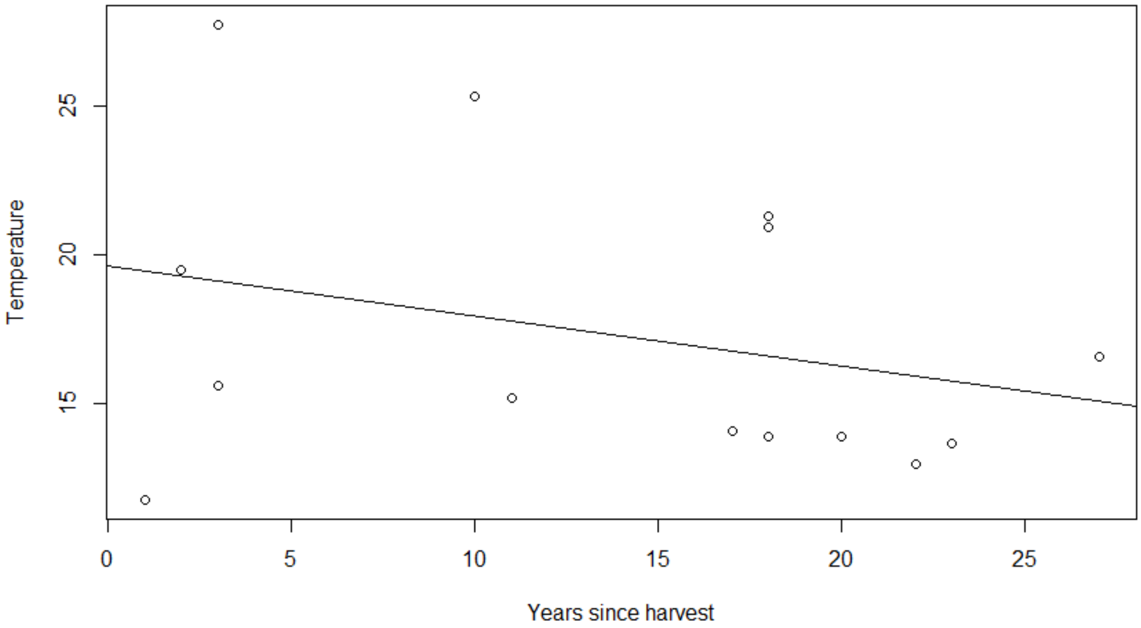

Fig. 5. Temperature plotted against years since harvest for the 14 sites sampled in the upper Waipaoa catchment, Gisborne, New Zealand in March 2019 ($F_{1,68}$=0.6.74 P=0.011, R^2=0.08).

Table 6. Analysis of Variance results testing for differences in physicochemical parameters between years since harvest for samples collected in March 2019 in the upper Waipaoa River catchment, Gisborne, New Zealand. Significant values are in bold.

	Years since harvest			
	d.f.	d.f.	F	P
Conductivity	1	10	0.671	**0.04**
Turbidity	1	10	0.014	0.91
Deposited Sediment	1	10	13.06	<0.01
Temperature	1	10	6.74	**0.011**
Habitat score	1	10	25.11	**<0.01**
pH	1	10	4.356	0.04
Dissolved Oxygen	1	10	1.068	0.30

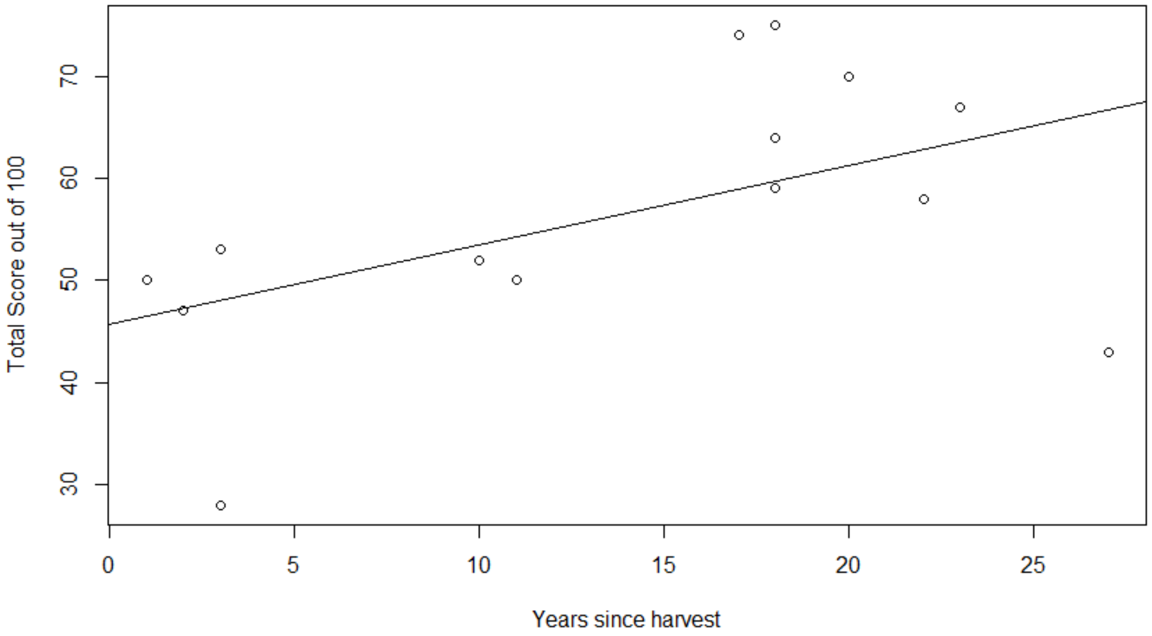

Fig. 6. Total habitat score plotted against years since harvest for the fourteen sites sampled in the upper Waipaoa catchment, Gisborne, New Zealand in March 2019 ($F_{1,68}$=25.11 P=<0.001, R^2=0.26).

Sediment

None of the four mean sediment particle sizes differed with years since harvest (Fig. 7, Table 7). The sediment grain sizes range from 1.62 to 10.3 ϕ indicating they are dominated by sandy mud or mud with varying coarseness of silt (Table 8). Cumulative frequency curves show particle size distributions are typically unimodal, bimodal and trimodal (Fig. 8). Site 4 had the largest sediment particle sizes and the only site with Leptokurtic Kurtosis, all other sites had either Mesokurtic or Platykurtic Kurtosis and were poorly sorted (1-2 ϕ) (Table 8).

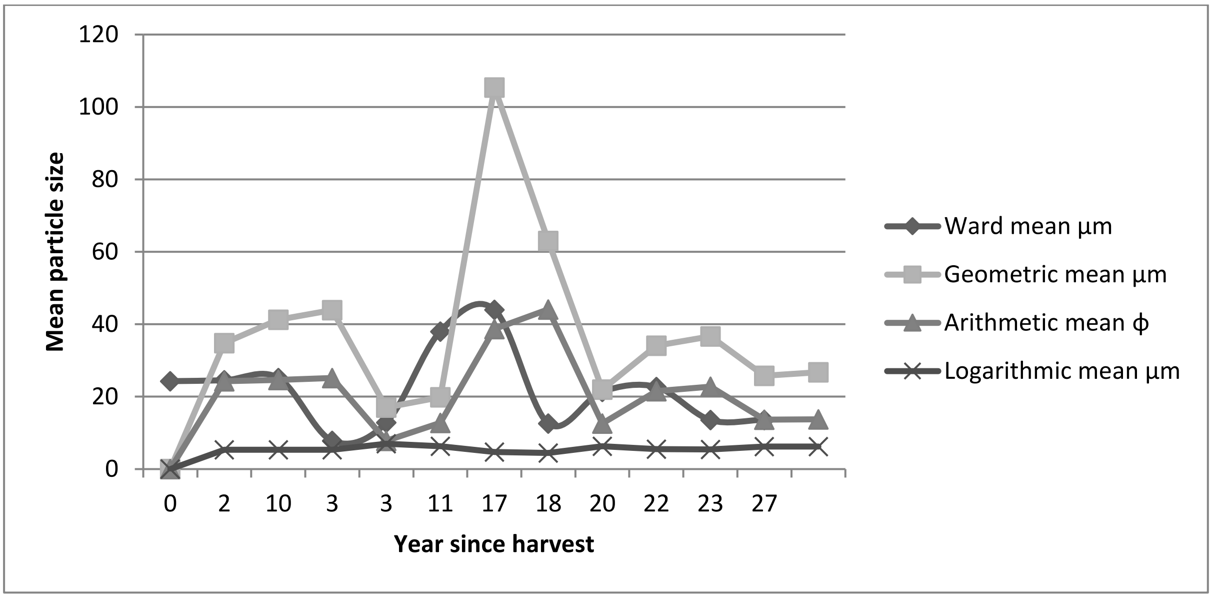

Fig. 7. Mean particle sediment size for twelve sites in the sampling in the Upper Waipaoa catchment, Gisborne

Table 7. Analysis of Variance results for sediment means for the twelve sites sampled in exotic forest in varying age classes in March 2019 in the upper Waipaoa catchment, Gisborne, New Zealand.

	Years since harvest			
Sediment mean	df	df	F	P
Logarithmic µm	1	10	0.001	0.974
Arithmetic φ	1	10	0.003	0.954
Geometric µm	1	10	0.005	0.946
Ward µm	1	10	0.005	0.945

Table 8. Textural group, sediment name, sample type and kurtosis of the twelve sites analysed by GRADISTAT for sediment particle size, sampled in March 2019, upper Waipaoa catchment, Gisbonre.

	Textural group	Sediment name	Sample type	Kurtosis
Site 0	Sandy Mud	Very Fine Sandy Coarse Silt	Unimodal, Poorly Sorted	Mesokurtic
Site 1	Sandy Mud	Very Fine Sandy Medium Silt	Bimodal, Poorly Sorted	Platykurtic
Site 2	Sandy Mud	Very Fine Sandy Medium Silt	Unimodal, Poorly Sorted	Mesokurtic
Site 3	Mud	Medium Silt	Trimodal, Poorly Sorted	Mesokurtic
Site 4	Mud	Medium Silt	Unimodal, Poorly Sorted	Leptokurtic
Site 5	Sandy Mud	Very Fine Sandy Coarse Silt	Bimodal, Poorly Sorted	Mesokurtic
Site 7	Sandy Mud	Very Fine Sandy Very Coarse Silt	Unimodal, Poorly Sorted	Mesokurtic
Site 8	Mud	Medium Silt	Bimodal, Poorly Sorted	Platykurtic
Site 9	Sandy Mud	Very Fine Sandy Medium Silt	Unimodal, Poorly Sorted	Mesokurtic
Site 10	Sandy Mud	Very Fine Sandy Medium Silt	Unimodal, Poorly Sorted	Mesokurtic
Site 11	Sandy Mud	Very Fine Sandy Coarse Silt	Trimodal, Poorly Sorted	Platykurtic
Site 12	Sandy Mud	Very Fine Sandy Medium Silt	Unimodal, Poorly Sorted	Mesokurtic

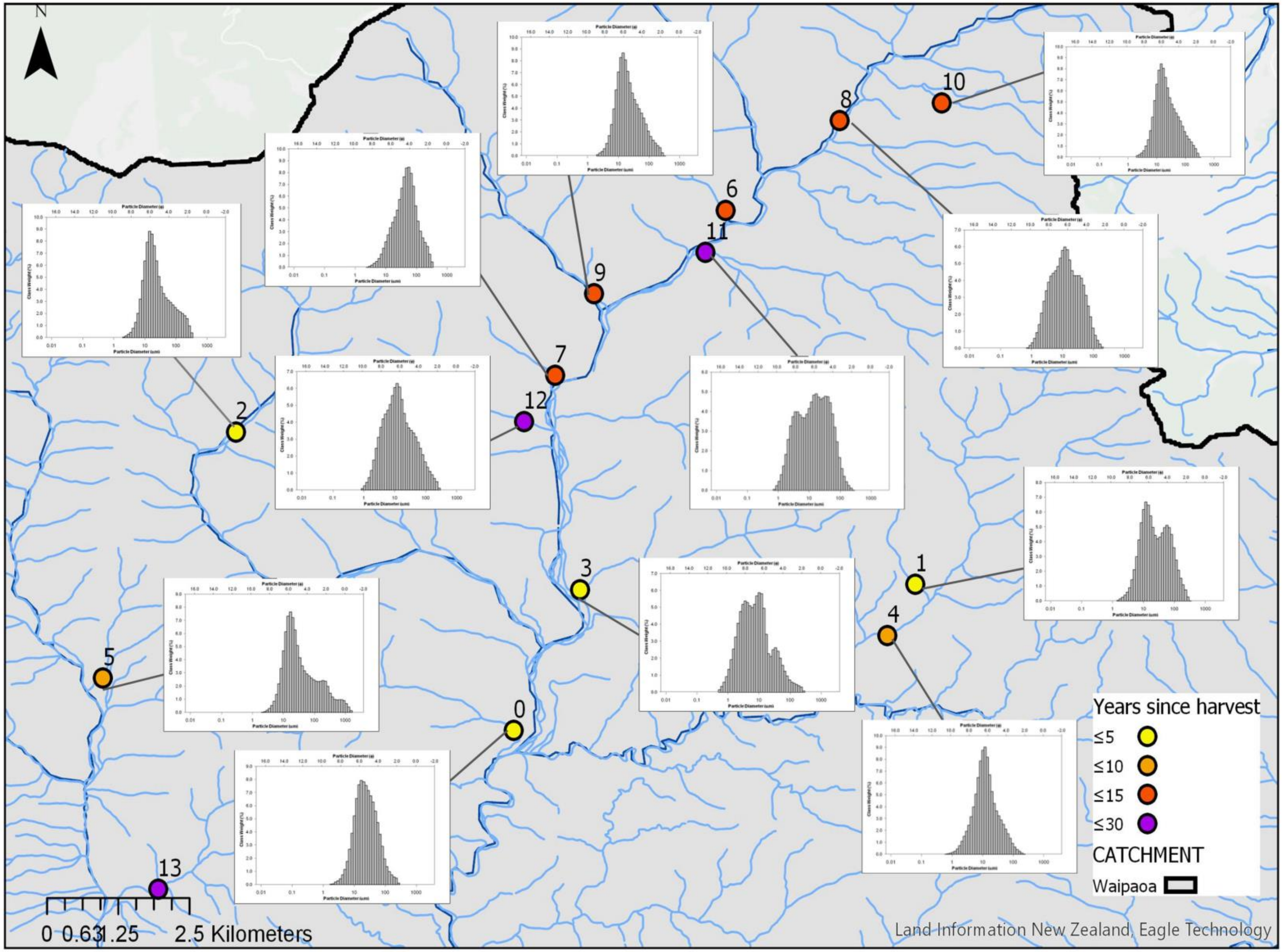

Fig. 8. Cumulative frequency curves of sediment particle diameters for twelve sites in different forest age classes in the Upper Waipaoa catchment, Gisborne.

Macroinvertebrates

A total of 99 macroinvertebrate taxa were collected from the 14 streams. The most diverse orders were Trichoptera (32 taxa), Diptera (24 taxa), Ephemeroptera (11 taxa), and Plecoptera and Coleoptera (9 taxa). The average number of taxa per sample was 15 and the total number of individuals per sample ranged from 3 to 674.

Coleoptera and Diptera dominated the samples in years one to three. From ten to 17 years, samples were dominated by Diptera, Trichoptera (*Hydrobiosis*) and Ephemeroptera (D*eleatidium*). From 18 to 27 years taxa that dominated were Diptera, Trichoptera, Ephemeroptera (*Austroclima* and C*oloburiscus*) (Fig. 9). Numbers of *Deleatidium* were 100 fold higher than any other taxa between the years 10-20. From 11 years following harvest, Trichoptera species increased from 20 species to 29 species.

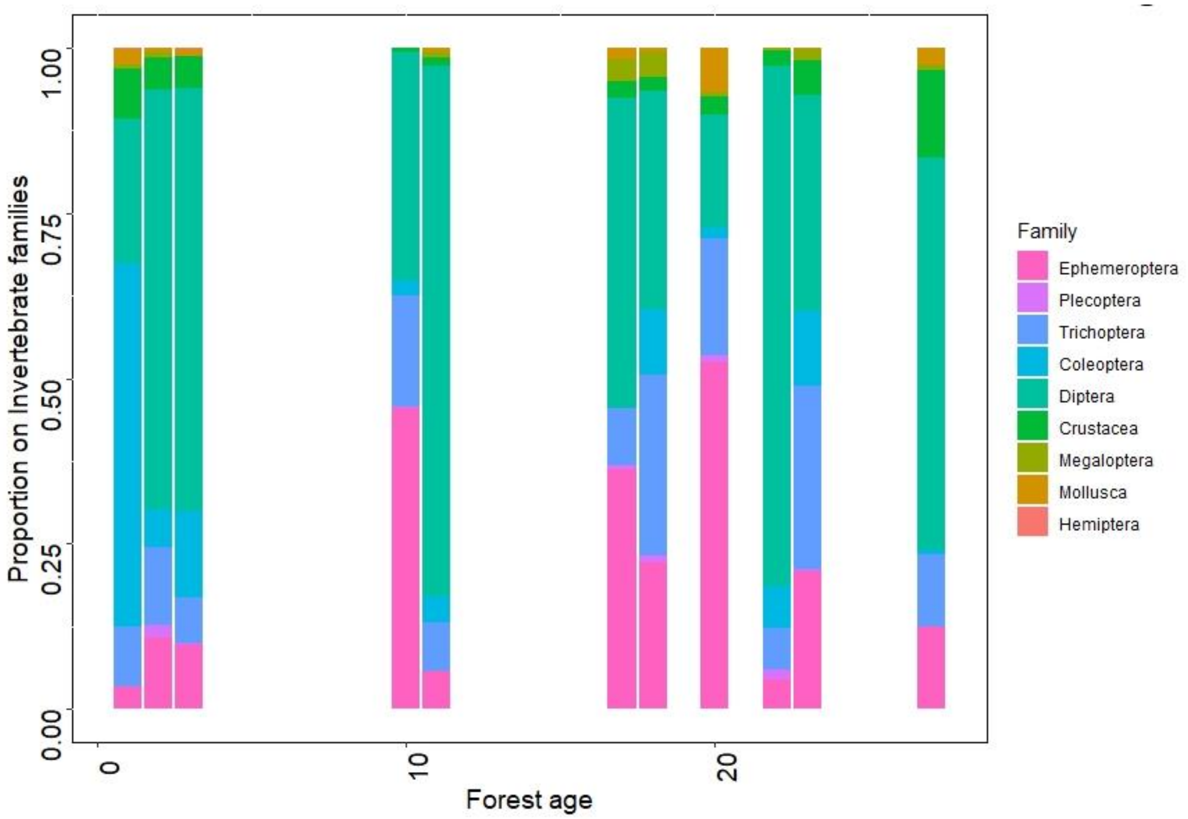

Fig. 9. Families of invertebrates from the fourteen sites sampled in different forest age classes from sites in the upper Waipaoa Catchment, Gisborne, New Zealand.

The QMCI ($F_{1,68}$=4.39 P=0.039, R^2=0.06) and %EPT($F_{1,68}$=14.08 P=<0.001 R^2=0.17) increased as the years since harvest increased. Deposited sediment decreased as years since harvest increased ($F_{1,68}$=7.462 P=0.008 R^2=0.09). EPT taxa ($F_{1,68}$=1.64 P=0.204, R^2=0.02) and total taxa had no significant change over the years since the forest was harvested (Fig. 10-Fig. 13).

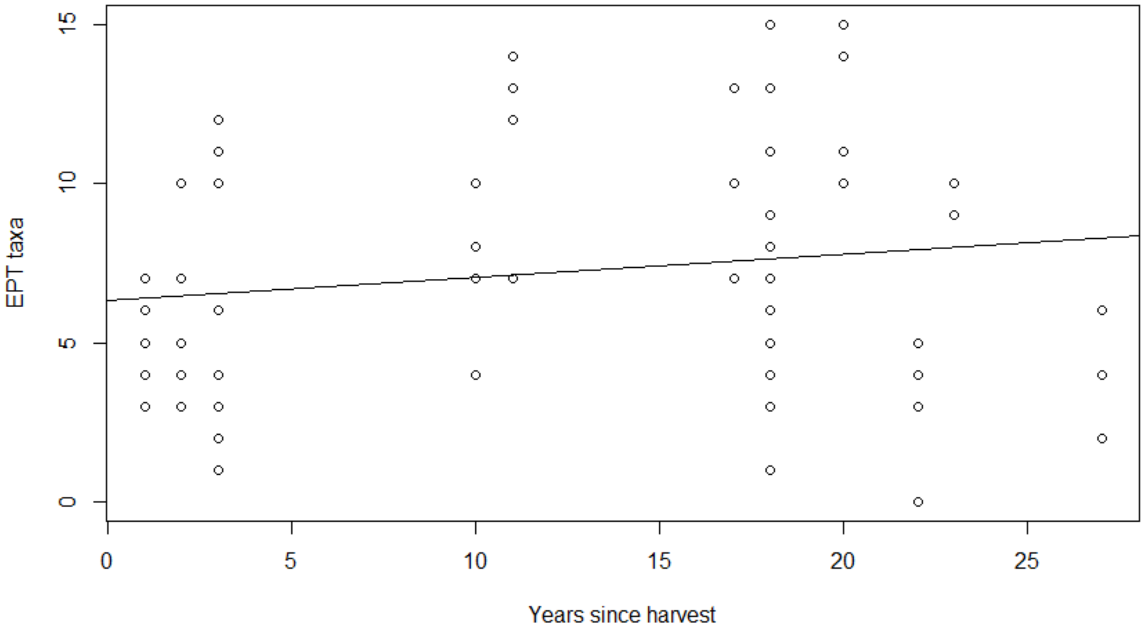

Fig. 10. EPT taxa plotted against years since forestry harvest for the upper Waipaoa catchment, Gisborne, sampled in March 2019. ($F_{1,68}$=1.64 P=0.204, R^2=0.02).

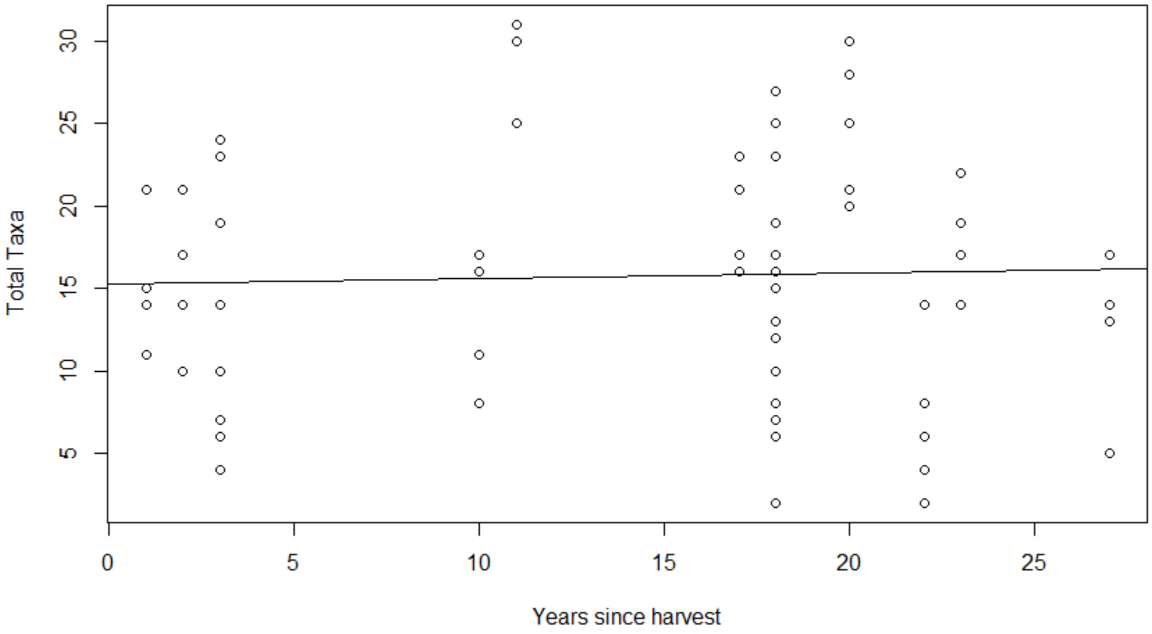

Fig. 11. Total taxa plotted against years since forestry harvest for the upper Waipaoa catchment, Gisborne, sampled in March 2019 ($F_{1,68}$=0.098 P=0.756, R^2=0.01).

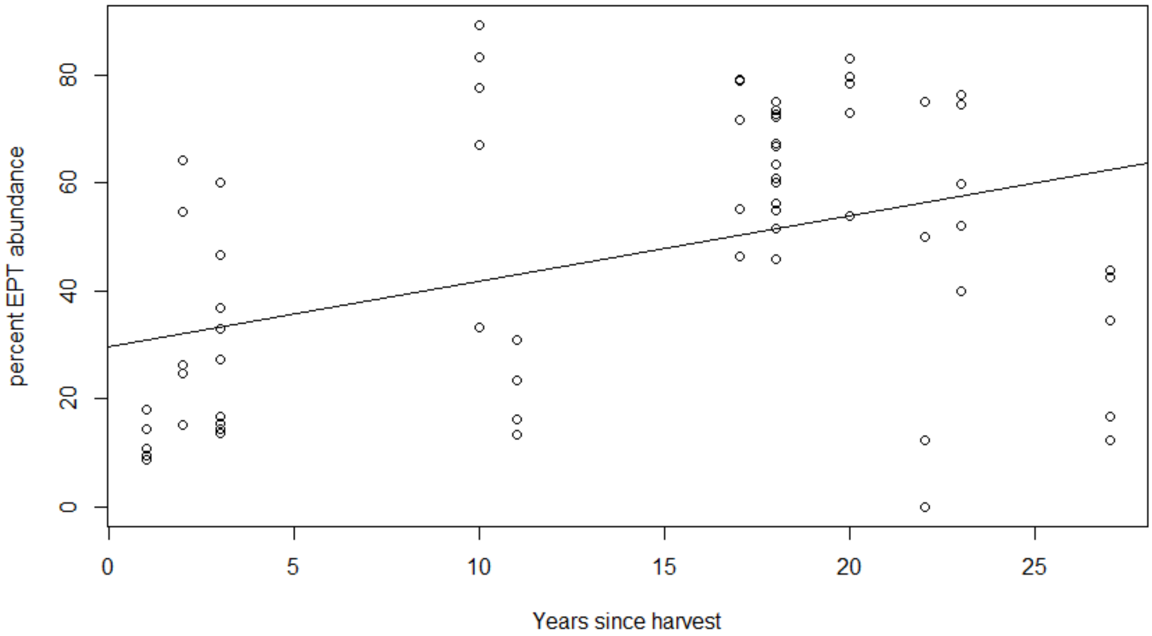

Fig. 12. %EPT abundance plotted against years since forestry harvest for the upper Waipaoa catchment, Gisborne, sampled in March 2019 ($F_{1,68}$=14.08 P=<0.001, R^2=0.17).

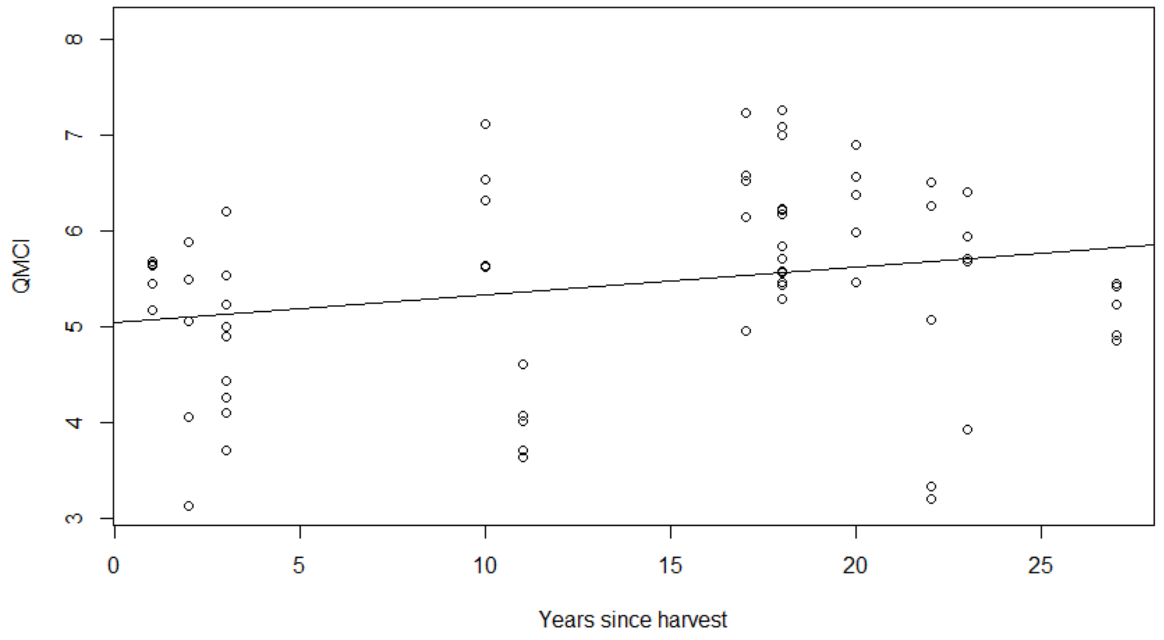

Fig. 13. QMCI plotted against years since forestry harvest for the upper Waipaoa catchment, Gisborne, sampled in March 2019 ($F_{1,68}$=4.39 P=0.039, R^2=0.06).

There was no relationship between QMCI and mean sediment particle size (Table 9). Temperature had no influence on QMCI ($F_{1,68}$=0.23 P=0.62, R^2=0.003) or %EPT ($F_{1,68}$=0.42 P=0.51, R^2=0.006).

Table 9. Analysis of Variance results for QMCI and particle size results from GRADISTAT for 12 sites sampled in the upper Waipaoa catchment, Gisborne, New Zealand. Significant values are in bold.

	QMCI			
Sediment	df	df	F	P
Logarithmic µm	1	58	1.987	0.164
Arithmetic φ	1	58	1.35	0.25
Geometric µm	1	58	1.33	0.273
Ward µm	1	58	1.48	0.228

The NMDS had a stress level of 0.09 indicating a significant repeatable pattern. Of the thirty two environmental variables measured, habitat parameters including riparian shade, bank vegetation, hydraulic heterogeneity, and the years since harvest had the most influence on macroinvertebrate community composition (Fig. 14).

Turbidity, mean sediment particle size, small gravel and bank erosion all had the strongest influence on communities that were in the 1-3 years since harvest period with the exception of one site which was 22 years since harvest.

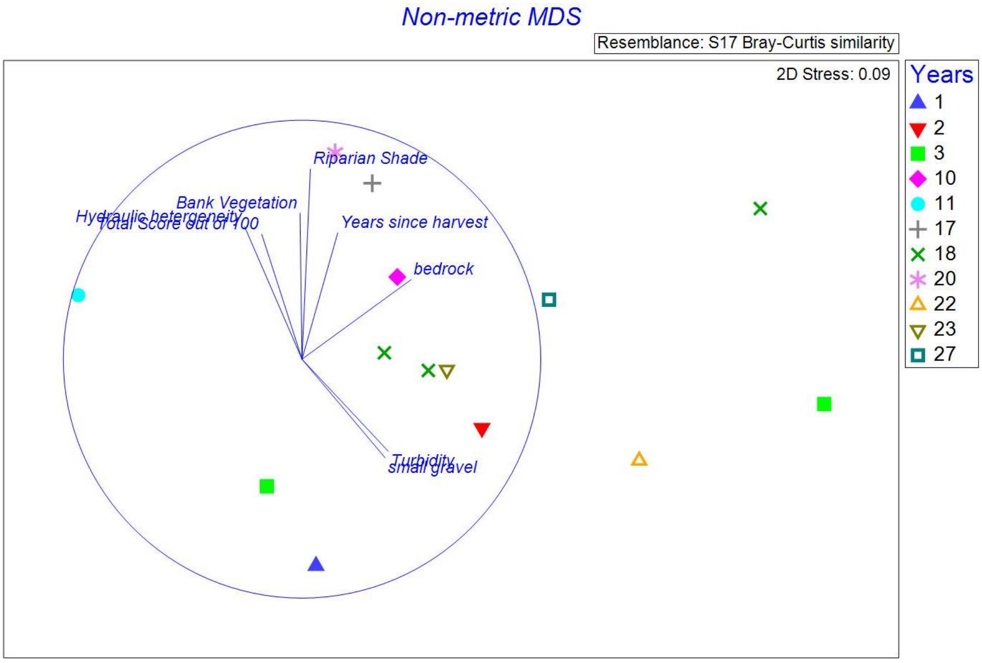

Fig. 14. NMDS of invertebrate communities collected at fourteen sites from the upper Waipaoa catchment in Gisborne in March 2019. Significant environmental variables are overlaid.

Discussion

Conductivities in East Coast streams are elevated compared to other areas (New Zealand average conductivity is *c*. 85µS/cm) particularly in turbid rivers and where catchments have high rates of erosion (Parkyn *et al*., 2006). The high conductivity levels (222-1034 µS/cm) in this study reflect the soft erodible geology. Conductivity values decreased as forest matured probably as there was less exposed sediment as vegetation matures stabilising sediment migration.

The abundance of sensitive EPT taxa was higher in streams harvested longer ago however; the total number of taxa did not change with the harvest cycle. Macroinvertebrate communities in streams that had been harvested ten years ago had more Trichoptera and Ephemeroptera than the streams that were more recently harvested, which were dominated by Diptera and Coleoptera, taxa more tolerant of degraded conditions. The change in community composition following forestry harvest in small streams is consistent with other studies that showed a similar response of a change in sensitive taxa (Death *et* a*l*., 2003; Baille *et* a*l*., 2005; Reid *et* a*l*., 2010). Reid *et* a*l*., (2010) found in small Coromandel streams following forest harvest, densities of tolerant taxa Diptera, Mollusca and Oligochaetes increased and EPT densities decreased. Reid *et* a*l*., (2010) also found that although the community structure was similar to pre-harvest, after 8 years the continual presence of high densities of Diptera was due to the persistence of fine sediment. In this study the densities of EPT individuals increased with forest maturity, however Diptera densities remained high at sites up to the 27 years following harvest. The presence of similar sediment particle sizes across the sites could explain the abundance of the sediment tolerant taxa at all sites.

Sites in the Pakuratahi catchment in Hawkes Bay had a decrease in QMCI following harvest where stream communities were dominated by Chironomidae, *Aoteapsyche* sp., Elmidae, and P. *antipodarum* (Death *et al*., 2003). Small streams in the Coromandel had a decline in QMCI from a range of sensitive taxa to invertebrates tolerant to pollution (Reid *et* a*l*., 2010). In both studies QMCI increased as the forest matured. This was also the case in the Upper Waipaoa catchment forest sites. No sites that were harvested less than ten years ago had excellent stream health based on their QMCI metric.

Thompson *et al*., (2009) found that the stream invertebrate communities that had the least change from before to after forest harvest were streams where riparian buffers were retained and stream shading was created over the harvested channel. Riparian vegetation is an important regulator of stream shade; it influences the thermal regime by reducing light and therefore solar energy and regulates stream temperature (Quinn and Wright Stow, 2008). Reductions of the effects of increased light following clear-fell harvest can be reduced by 40% when an indigenous riparian buffer remains (Boothroyd *et al*., 2004). The core drivers for macroinvertebrate communities in the small streams in the Upper Waipaoa catchment in exotic forest that had been established for more than ten years were habitat characteristics including; riparian shade, bank vegetation, stream hydraulics and overall habitat score. Indigenous riparian vegetation in forestry areas have proven to have important functions for stream ecosystems including; controlling the levels of organic inputs including sediment and logging debris, limiting access of forestry machinery to the stream and providing a buffer for contaminants, reducing light levels and therefore primary production and algal biomass, increasing organic inputs including leaf litter and woody material, providing terrestrial habitat for adult invertebrates and decreasing stream bank erosion and reducing changes in stream morphology (Clinnick, 1985; Harding *et al*., 2000; Boothroyd *et al*., 2004, Quinn *et al*., 2004). Macroinvertebrate communities in forest harvest areas in the Coromandel were found to be similar to invertebrate communities in indigenous forest when riparian buffers remained intact following harvest (Quinn *et al*., 2004). Davies and Nelson (1994) found in eucalyptus forest, effects on stream health including sediment inputs and temperature were dependant on buffer width rather than erodibility or degree of slope. The retention of a 10 m riparian strip in harvest areas reduces effects on macroinvertebrate communities (Thompson *et* al., 2009), however to protect streams from the effects of forestry harvest a riparian margin of 30 m is required (Clinnick, 1985).

The stressors that had the most influence on the communities for the first three years following harvest were turbidity, sediment size, bank erosion and sediment particle size. These stressors are directly related to the level of environmental disturbance after harvest in the upper Waipaoa catchment with a reduction in riparian vegetation, influx of sediment resulting in a decrease in substrate size and increased exposed sediment on slopes influencing turbidity and conductivity levels. The exception to this was a site that was 22 years old after forest harvest; this site had active gully erosion upstream that was continuously depositing sediment into the stream. These inputs caused similar effects to sites where harvest was completed in the last three years. These findings suggest that if buffer strips and riparian vegetation were retained next to small streams in the upper Waipaoa catchment as part of the harvesting process, the effects on macroinvertebrate communities and stream health would be reduced.

The combination of increased surface runoff and exposed sediment from clear-fell harvest results in more sediment and a reduced substrate size in streams (Harding *et al*., 2000; Wood and Fahey, 2006; Quinn *et* a*l*., 2009). The movement of sediment in the forest area following harvest is predominantly driven by storm and rainfall events, these events can both flush sediment from streams and assist in post-harvest stream recovery (Death *et* a*l*., 2003; Marden *et* a*l*., 2005). The results of the fine sediment particle sizes in the upper Waipaoa catchment were similar across all sites sampled with higher densities of deposited sediment and silt in streams following forest harvest. The similar particle sizes could be a result of continual inputs of, or residual sediment from forest harvest that is yet to be transported through the stream network. Mean particle size along with bank erosion were two of the strongest drivers of macroinvertebrate communities in forest harvested less than four years ago. This indicates that environmental stressors that recover over time, such as riparian vegetation, shade and habitat could be more important for faster macroinvertebrate recovery than the particle size and density of fine sediment. The continual presence of fine sediment throughout the harvest cycle, as was the case in this study has the potential to inhibit macroinvertebrate communities preventing full recovery following harvest. The fine sediment provides habitat for less sensitive taxa and allows taxa such as Diptera to dominate communities for the whole harvest period. The legacy issue of fine sediment remaining instream has the potential to be compounded as forest is harvested repeatedly and sediment inputs continue. Persistent impacts such as continual fine sediment inputs and lack of organic inputs such as woody debris, means it is possible that streams may improve in macroinvertebrate density and diversity but may not fully recover in the 25-30 year rotation period (Reid *et* a*l*., 2010). This could explain why the number of taxa in the upper Waipaoa River catchment did not change over the rotational cycle and the few sensitive Plecoptera taxa found at sites. Residual sediment from the first rotation of harvesting and previous land use may be present at mature forest sites and although the density of sensitive EPT taxa increased over time, residual fine sediment could be influencing species diversity.

Some parameters changed over time following harvest. The rate of sedimentation, rainfall, recolonisation of invertebrates and rate of vegetation reestablishment will all be factors influencing stream health since harvesting time (Death *et* a*l*., 2003; Quinn *et* a*l*., 2004). It is difficult to understand exactly what features improve at what time, as there are many other determining influences on macroinvertebrate communities. However the predominant finding in this study is that over the time following harvest, macroinvertebrate communities do improve, with higher densities of sensitive taxa driven by the establishment of riparian vegetation and therefore further habitat improvements. The recovery rate of the streams in soft geology in Gisborne is similar to streams following forestry harvest in Hawke’s Bay and Coromandel, with macroinvertebrate communities having higher densities of sensitive EPT taxa ten years following harvest (Death *et* a*l*., 2003, Reid *et* a*l*., 2010).

In summary, the influences of forest harvest in soft geology on macroinvertebrate communities in streams in the Upper Waipaoa catchment appear to be similar to harvest impacts in other regions with different geology types. Turbidity, sediment density and size of sediment influence macroinvertebrate communities for up to ten years following forest harvest until riparian vegetation has established. As riparian vegetation establishes and habitat characteristics improve, macroinvertebrate communities include higher densities of sensitive EPT taxa. Residual fine sediment remains over the whole harvest cycle which could be inhibiting the complete recovery on macroinvertebrate communities. To reduce effects of forestry harvest, established continuous riparian buffers that remain during the whole harvest cycle will ensure macroinvertebrate communities and stream habitats are least affected.

References

Black, R.D., (1977). Rivers of change; early history of the upper Waipaoa and Mangatu catchments. Unpublished Report, New Zealand Forest,Service, Forest Research Institute, Christchurch, New Zealand.

Blott, S.J., Pye, K., (2001). Gradistat: a grain size distribution and statistics package for the analysis of unconsolidated sediments. Earth Surface Processes and Landforms 26, (1237 – 1248).

Boothroyd, I.J., Quinn, J.M., Costley, K.J., Langer, E.R., Steward, G., (2004). Riparian buffers mitigate effects of pine plantation logging on New Zealand streams. 1. Riparian vegetation structure, stream geomorphology and periphyton. Forest Ecology and Management 191:129-146.

Clinnick, P.F., (1985). Buffer strip management in forest operations: a review. Australian Forestry 48, 34–45.

Davies, P. E.; Nelson, M. (1994). Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research 45: 1289-1305.

Death, R., Baillie, B., and Fransen, P. (2003). Effect of Pinus radiata logging on stream invertebrate communities in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research. 37. 507-520. 10.1080/00288330.2003.9517185.

Fuller, I.C., Marden,M (2011). Slope–channel coupling in steepland terrain: Afield-based conceptual model from the Tarndale gully and fan, Waipaoa catchment, New Zealand. Geomorphology 128 (105-115).

Glade, T., 1998. Establishing the frequency and magnitude of landslide-triggering rainstorm events in New Zealand. Environmental Geology 35, 160 – 174.

Hamilton, D., Kelman, E.H.H., (1952). Soil conservation survey of the Waipaoa River catchment, Poverty Bay, New Zealand. Unpublished Report, New Zealand Soil Conservation and Rivers Control Council, Wellington, New Zealand.

Harding, J. S.; Quinn, J. M.; Hickey, C. W. (2000). Effects of mining and production forestry. In: Collier, K. C.; Winterbourn, M. J. ed. New Zealand stream invertebrates: ecology and implications for management. Christchurch, New Zealand Limnological Society. Pp. 230-259.

Hicks, D.M., Gomez, B., Trustrum, N.A., (2000). Erosion thresholds and suspended sediment yields, Waipaoa River basin, New Zealand. Water Resources Research 36, 1129–1142.

Marden, M., Arnold, G., Gomez, B., Rowan, D., (2005). Pre- and post-reforestation gully development in Mangatu Forest, East Coast, North Island, New Zealand. River Research and Applications 21, 757–771.

Marden,M., Arnold,G., Seymour, A.,(2012). Rambling History and distribution of steepland gullies in response to land use change, East Coast region, North Island, New Zealand Geomorphology, 153 (2012), pp. 81-90.

Marden, M., Betts, H., Arnold, G., Hambling, R., (2008). Gully erosion and sediment load:Waipaoa, Waiapu and Uawa Rivers, eastern North Island, New Zealand. SedimentDynamics in Changing Environments: In: Schmidt, J., Cochrane, T., Phillips, C.,Elliott, S., Davies, T., Basher, L. (Eds.), Proceedings Christchurch Symposium, 325.IAHS Publ, Wallingford, UK, pp. 339–350.

Marden.M, Herzig. A, Basher.L (2014). Erosion process contribution to sediment yield before and after the establishment of exotic forest: Waipaoa catchment, New Zealand. Geomorphology, 226, 162-174.

McLellan, I. D. (1999). A revision of Zelandoperla Tillyard (Plecoptera: Gripopterygidae: Zelandoperlinae). New Zealand Journal of Zoology, 26(3), 199-219.

Noel, D.S., Martin, C.W., Federer, C.A., (1986). Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environmental Management 10 (5),661–670.

O'Loughlin, C.L., (1974). The effect of timber removal on the stability of forest soils.Journal of Hydrology. New Zealand 13, 121–134.

Parkyn, S.M.; Davies-Colley, R.; Scarsbrook, M.R.; Halliday, N.J.; Nagels, J.W.; Marden, M.; Rowan, D. (2006). Pine afforestation and stream health: a comparison of land-use in two soft rock catchments, East Cape, New Zealand. New Zealand Natural Sciences 31: 113-135.

Pearce, A.J., Black, R.D., Nelson, C.S., (1981). Lithologic and weathering influences on slopeform and process, eastern Raukumara Range, New Zealand. Erosion and SedimentTransport in Pacific Rim Steeplands. : In: Davies, T.R.H., Pearce, A.J. (Eds.), Proceedings Christchurch Symposium, 132. IAHS Publication, Wallingford, UK,pp. 95–122.

Piégay, H., Walling, D.E., Landon, N., He, Q., Liébault, F., Petiot, R., (2004). Contemporary changes in sediment yield in an alpine mountain basin due to afforestation (the upper Drôme in France). Catena 55, 183–212.

Quinn, J.M., Boothroyd, I.K.G., Smith, B.J., (2004). Riparian buffers mitigate effects of pine plantation logging on New Zealand streams. 2. Invertebrate communities. Forest Ecology Management 191, 199-213.

Quinn, J.M., Croker, G.F., Smith, B.J., Bellingham, M.A., (2009). Integrated catchment management effects on runoff, habitat, instream vegetation and macroinvertebrates in Waikato, New Zealand, hill country streams. New Zealand Journal of Marine and Freshwater Research, 43(3):775-802.

Quinn, J. M.; Halliday, J. (1999). Effects of forestry and harvesting on Tairua Forest stream habitat and invertebrates. Sixth annual report incorporating the results from 1993-1999. Client report CHH00202/1, NIWA, Hamilton.

Quinn J.M., and Wright-Stow A.E. (2008). Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest ecology and management 256:2101-2109.

Reid,D.J., Quinn, J.M., Wright-Stow, A. E. (2010). Responses of stream macroinvertebrate communities to progressive forest harvesting: Influences of harvest intensity, stream size and riparian buffers. Forest Ecology and Management, 260, 1804-1815.

Reid, L.M., Page, M.J., (2003). Magnitude and frequency of landsliding in a large New Zealand catchment. Geomorphology 49, 71 – 88.

Silsbee, D.G., Larson, G.L., (1983). A comparison of streams in logged and unlogged areas of Great Smokey Mountains National Park. Hydrobiologia 102, 99–111.

Thompson, R., Phillips, N., and Townsend, C.R. (2009). Biological Consequences of Clear-Cut Logging Around Streams – Moderating Effects of Management. Forest Ecology and Management. 257. 931-940. 10.1016/j.foreco.2008.10.025.

Winterbourn M.J., Gregson K.L. and Dolphin C.H. (2006). Guide to the aquatic insects of New Zealand, Entomological Society of New Zealand Auckland.

Winterbourn, M. J. (1973). A guide to the freshwater Mollusca of New Zealand. Tuatara, 20(3), 141-159.

Wood, G., Fahey, B. (2006). Chapter 4 Forestry Effects on Stream Flow and Water Yield, in Eyels G and Fahey B eds. The Pahuratahi Land Use Study: A 12-year paired catchment study of environmental effects of pinus radiata forestry: Napier, Hawkes Bay Regional Council. Pp 31-49.

Concluding summary

New Zealand historically was covered in indigenous forest that has since been cleared primarily for agriculture and forestry (McGlone, 1989). The original forest cover would have provided freshwater habitat favouring cool, shaded stable streams that supported a diverse assemblage of invertebrates and fish that were adapted to forest stream conditions. The change in land use from indigenous forest, to land that produces food and fibre, has resulted in a change to stream networks, stream headwaters, food resources, sedimentation, habitat, less wood and more deposited sediment (Death *et al.*, 2003; Quinn and Wright-Stow, 2008; Clapcott *et al.*, 2011). Following land use change stream habitat is more homogenised, flows are less stable, flooding is more frequent and stream morphology is changed (Reid *et al.*, 2010; Newson, 2000).

The findings from the previous two chapters have highlighted that in the East Coast Region land use influences freshwater invertebrate communities in both soft and hard geology. Macroinvertebrate community diversity and biotic metrics were reduced as sediment inputs increased with intensification of land use. Julian *et al.*, (2017) determined relationships between land use intensity and water quality were complex with many factors such as soil type, livestock density, slope and disturbance influencing results. In particular the grazing and land use practice involved for cattle (both dairy and beef cattle) have significant impacts on freshwater systems through increased runoff, nutrient inputs and their requirement of high producing pastures. Water quality parameters that are linked to geology and sedimentation rates were the main drivers of macroinvertebrate communities in the land use types in the Gisborne region. Stream conductivity determined by both land use and geology, influenced freshwater communities in both studies.

The increasing export and global demand for New Zealand dairy products and increasing agricultural production could mean further reductions is sheep numbers, increases in cattle and further demand for production of land (Stats NZ, 2018). The legacy and lag effect of nutrients diffusely entering waterways in New Zealand, means that the already degraded rivers, coupled with continued land use change and intensity, will continue to have pressures from land use, potentially for decades (Howard-Williams *et al.,* 2010; Julian *et al.*, 2017). The effect on water quality is amplified in areas where soil type is more erodible or free draining (Julian *et al*., 2017). Independent influences on freshwater communities of land use and geology highlight the importance of considering geology when making decisions regarding land use.

The harvesting of production forests is expected to rise by 40% over the next decade and substantially more in the following three decades due to the establishment of forests under the 1 Billion Trees project (MPI, 2013; Te Uru Rakau 2019). Harvest planning needs to be considered at the afforestation stage to ensure the effects on freshwater systems and erosion potential is mitigated from the beginning of the project. Planting exotic forest assists the control of erosion (Marden *et al*., 2014) however the harvesting of the trees leaves the most erosion prone, steep slopes exposed as reestablishment of vegetation takes several years (Fahey *et al*., 2003). The general recovery time for streams in soft geology in areas of forest harvest was a minimum of ten years. This was driven predominantly by riparian vegetation, stream hydraulics and increased habitat that favoured more sensitive EPT taxa and a reduction in pollution tolerant taxa. The continual presence of high densities of pollution tolerant Diptera throughout the rotation cycle of plantation forest suggests sediment inputs are still having an influence on community composition. This could be a response to the lithology and other sediment sources in the landscape rather than solely forest harvest.

Chapter 2 results found turbidity, sediment density and size of sediment influences macroinvertebrate communities for up to ten years following forest harvest in soft geology until riparian vegetation has established and inputs of sediment reduced. The establishment of riparian buffers and the ability of vegetation to control sediment inputs and therefore the thermal regime of a stream is more important than the surrounding slope or erodibility of land (Davies, 1994). Continuous riparian strips are more effective at providing protection from logging activities compared to fragmented riparian vegetation, and can support the same macroinvertebrate communities as indigenous forested streams (Quinn *et el*., 2004). To minimise negative effects on ecosystems, riparian buffers should be implemented to 30 m to prevent the negative impacts on waterways (Clinnick, 1985). Large areas of river length are surrounded by exotic forest in New Zealand, and if indigenous vegetation formed corridors providing buffers, it would not only improve stream health, but will enhance terrestrial habitat for flora and fauna and improve biodiversity.

Climate change and the potential for more frequent and severe storm events, interspersed with more extreme dry conditions causing droughts (MfE, 2018), coupled with existing areas of plantation forestry and agricultural land use could result in increases in the detrimental effects of land use practice on freshwater ecosystems in the future. It is likely that reduction in stream flows during droughts associated with forecast climate change and the increases in sediment from storms will result in a higher benthic load of deposited sediment in freshwater systems nationally (Tisseuil *et al*., 2012, Comte *et al*., 2013). The results from chapters 1 and 2 highlight the independent influences that geology and land use have on freshwater communities. The national decline in freshwater quality has evoked changes in legislation and increased resources into stream restoration and rehabilitation (MfE, 2014). It is therefore important that geological influences are considered when making decisions on land use change and the potential effects on freshwater ecosystems.

References

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

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

Clinnick, P.F., (1985). Buffer strip management in forest operations: a review. Australian Forestry 48, 34–45.

Comte, L., L. Buisson, M. Daufresne, and G. Grenouillet. (2013). Climate-induced changes in the distribution of freshwater fish: observed and predicted trends: Climate change and freshwater fish. Freshwater Biology 58:625-639.

Davies, P. E. and Nelson, M. (1994). Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research 45: 1289-1305.

Death, R., Baillie, B., and Fransen, P. (2003). Effect of Pinus radiata logging on stream invertebrate communities in Hawke's Bay, New Zealand. New Zealand Journal of Marine and Freshwater Research. 37. 507-520. 10.1080/00288330.2003.9517185.

Fahey, B., Marden, M., and Phillips, C. (2003). Sediment yields from plantation forestry and pastoral farming, coastal Hawke's Bay, North Island, New Zealand. Journal of Hydrology (New Zealand), *42*(1), 27-38.

Marden, M., Herzig. A., Basher. L. (2014). Erosion process contribution to sediment yield before and after the establishment of exotic forest: Waipaoa catchment, New Zealand. Geomorphology, 226, 162-174.

McGlone, M. (1989). The Polynesian settlement of New Zealand in relation to environmental and biotic changes:New Zealand Journal of Ecology 12:116-129

Ministry for the Environment. (2014). National Policy Statement for Freshwater Management 2014. New Zealand Government.nz

Ministry for the Environment. (2018). Climate change predictions for the Gisborne and Hawke’s Bay regions. Retrieved from: https://www.mfe.govt.nz/climate-change/likely-impacts-of-climate-change/how-could-climate-change-affect-my-region/gisborne.

Newson, M. D., and Newson. C. L. (2000). Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. Progress in Physical Geography 24:1195- 1217.

Quinn, J.M., and Wright-Stow, A.E. (2008). Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest ecology and management 256:2101-2109.

Reid, D.J., Quinn, J.M., Wright-Stow, A.E. (2010). Responses of stream macroinvertebrate communities to progressive forest harvesting: Influences of harvest intensity, stream size and riparian buffers. Forest Ecology and Management, 260, 1804-1815.

Statistics New Zealand. (2018). New Zealand Landcover. Published April 2019. www.stats.govt.nz.

Te Uru Rakau. (2019). One Billion Trees Fund: Six month monitoring and evaluation report. A monitoring and evaluation report on the state of the One Billion Trees Fund to 30 June 2019. New Zealand Government.nz

Thompson, R., Phillips, N., and Townsend, C.R. (2009). Biological Consequences of Clear-Cut Logging Around Streams – Moderating Effects of Management. Forest Ecology and Management. 257. 931-940. 10.1016/j.foreco.2008.10.025.

Ministry of Primary Industries. (2013). Situation and outlook for primary industries, 2013. Ministry of Primary Industries, New Zealand.

Appendix 1.

Site names, locations, land use and geology type for the 79 sites sampled in the Gisborne region from 2016-2018.

Site id	Site name	NZTM	NZTM	Land use	Geolo
GRES 1	Kopuapunamu Str Trib at Te	2067753	5815210	Pasture	Soft
GRES 2	Waipaoa Trib at Lavenham-	2026228	5724270	Pasture	Soft
GRES 3	Parihihonou Str at SH2	2003439	5734503	Pasture	Soft
GRES 6	Waiau River at Tauwhareparae	2048998	5757668	Exotic forest	Soft
GRES 7	Makarika Str at Keelan Rd	2058304	5781596	Exotic forest	Hard
GRES 8	Mangatu Trib	2006125	5723208	Exotic forest	Hard
GRES 9	Huitatariki Str	2041996	5800444	Exotic forest	Hard
GRES 10	Waipiro Str at Te Puia	2068024	5778597	Pasture	Soft
GRES 11	Kouetumara Str at Ihungia Rd	2054357	5777065	Exotic forest	Soft
GRES 13	Makokomuka Str at Anaura Rd	2062586	5763283	Pasture	Soft
GRES 14	Anaura Str at Anaura Bay Rd	2065929	5756241	Pasture	Soft
GRES 15	Waipaoa at Armstrong Rd	2024364	5751596	Exotic forest	Hard
GRES 16	Whakauranga Br at West Ho Rd	2048353	5740873	Exotic forest	Soft
GRES 17	Waikohu Rv at Oliver Rd	2000375	5737651	Pasture	Soft
GRES 18	Kaitawa Str at Wharf Rd	2064142	5737658	Pasture	Soft
GRES 19	Waikohu trib at Whakarau Road	2014917	5734666	Pasture	Soft
GRES 20	Waihuka River at No.3 Br	2014917	5734666	Pasture	Soft
GRES 21	Waikohu River at No.3 Br	2017156	5732909	Pasture	Soft
GRES 22	Kurunui Str at Holdsworth Br	2018776	5732185	Pasture	Soft
GRES 23	Waihuka at No2 Br	2007056	5731778	Pasture	Soft
GRES 24	Mangaoai Str at Mangaoai Rd	2028744	5732006	Exotic forest	Soft
GRES 25	Pakarae Trib at Whangara Rd	2057804	5728620	Pasture	Soft
GRES 26	Te Arai Trib at Waingake Rd	2017015	5697252	Exotic forest	Soft
GRES 27	Pakarae trib Stevens Road	2053366	5727936	Pasture	Soft
GRES 28	Wharekopae above falls	2001782	5724646	Pasture	Soft
GRES 29	Waikakariki Trib at Quarry	2020169	5712404	Pasture	Soft
GRES 30	Maraetaha Str at No2 Br	2025948	5693812	Pasture	Soft
GRES 31	Haupapa Str at Tahora	1985744	5725142	Pasture	Hard
GRES 32	Mangaotara Str at Tiniroto Rd	1994134	5695106	Pasture	Soft
GRES 33	Upper Motu Trib at Mangatu	2004775	5761849	Indigenous forest	Hard
GRES 34	Marumoko Str at Marumoko Rd	2001854	5760106	Pasture	Hard
GRES 36	Koranga Trib at Rakauroa Rd	1988345	5731989	Indigenous forest	Hard
GRES 37	Lottin Point Road Stream	2055511	5830469	Pasture	Hard
GRES 38	Karakatuwhero Trib at	2069226	5819930	Exotic forest	Soft
GRES 39	Whangaparoa Trib at Waikura	2046733	5818704	Pasture	Hard
GRES 40	Mangaoparo River at	2061952	5802630	Exotic forest	Hard
GRES 41	Waitekaha Str at Tuparoa Rd	2073004	5790444	Pasture	Soft
GRES 42	Mangaehu Stream at Marshall	2038730	5725069	Pasture	Hard
GRES 45	Te Arai Trib at Waugh Rd	2020695	5701503	Pasture	Soft
GRES 46	Te Arai Rv at Waingake	2016364	5695803	Pasture	Soft

GRES 47	Mangakino Str at Mangatokerau	2059325	5748333	Indigenous forest	Soft
GRES 48	Koranga Rv at Koranga Valley	1985760	5733698	Pasture	Hard
GRES 49	Papokeka Str at Pehiri	1995205	5719251	Pasture	Soft
GRES 50	Whakarau Trib at Whakarau Rd	1999503	5749523	Pasture	Soft
GRES 51	Motu River at Kotare Station	1993092	5744473	Pasture	Hard
GRES 52	Motu River @ Matawai	1989481	5737668	Indigenous forest	Soft
GRES 53	Wharekahika River U/S of	2066639	5828585	Pasture	Soft
GRES 54	Mangatutu Str at Sh35-	2064525	5825633	Indigenous forest	Hard
GRES 55	Oweka River at SH35 Bridge	2055023	5826877	Pasture	Soft
GRES 56	Karakatuwhero River at SH35	2070952	5822227	Indigenous forest	Hard
GRES 57	Awatere River at SH35 Bridge	2071787	5816778	Exotic forest	Hard
GRES 58	Waiapu River at Rotokautuku	2065726	5792775	Pasture	Hard
GRES 59	Poroporo River at Rangitukia	2078060	5804077	Pasture	Soft
GRES 60	Mangaoporo River at Tutamatai	2065995	5798969	Pasture	Hard
GRES 61	Mata River at Pouturu Br	2049649	5776496	Pasture	Soft
GRES 62	Mata River at Aorangi	2057930	5787002	Pasture	Hard
GRES 63	Ihungia River at Ihungia Rd Br	2054344	5776298	Pasture	Soft
GRES 64	Hikuwai River at Willowflat	2060929	5748624	Pasture	Soft
GRES 65	Hikuwai River at No 4 Bridge	2060245	5758435	Pasture	Soft
GRES 66	Mangaheia River at Paroa Road	2058838	5742590	Pasture	Soft
GRES 67	Waihirere Str at Domain	2030997	5719154	Indigenous forest	Soft
GRES 70	Waimata River at Goodwins Rd	2041185	5711037	Pasture	Soft
GRES 71	Waimata River at Monowai	2039271	5724954	Pasture	Soft
GRES 72	Taruheru River at Tuckers Rd	2032521	5712114	Pasture	Soft
GRES 74	Waipaoa River at Kanakanaia	2028045	5710097	Pasture	Soft
GRES 75	Waipaoa River at Matawhero	2002197	5756583	Pasture	Soft
GRES 76	Motu River above Falls	2018693	5698857	Pasture	Soft
GRES 77	Te Arai River at Pykes Weir	2022843	5712887	Pasture	Soft
GRES 78	Whakaahu Str at Brunton Rd	2012301	5736256	Pasture	Soft
GRES 79	Waikohu River at Mahaki	2018759	5753108	Pasture	Soft
GRES 80	Wharekopae River at Rangimoe	2018802	5741290	Pasture	Soft
GRES 81	Mangatu River at Omapere	2030986	5742400	Pasture	Soft
GRES 82	Waingaromia River at Terrace	2058300	5723951	Pasture	Soft
GRES 83	Pakarae River at Pakarae	1997728	5742189	Pasture	Soft
GRES 84	Matawai Stream at Tawai	2041908	5717728	Pasture	Soft
GRES 85	Makahakaha Stream	2015825	5691611	Exotic forest	Soft
GRES 86	Te Arai River at DW Bush Intake Above Weir	2000500	5712798	Indigenous forest	Soft
GRES 89	Mata Upper	2028054	5773530	Indigenous forest	Hard
GRES 90	Mangaokura Stream	2035886	5781948	Indigenous forest	Hard

Appendix 2.

Photographs of sites sampled in March 2019, in the upper Waipaoa catchment, Gisborne.

Top left: Site 0, top right: Site 1, bottom left: Site 2, bottom right: Site 3.

Top left: Site 4, top right: Site 5, bottom left: Site 6, bottom right: Site 7.

Top left: Site 8, top right: Site 9, bottom left: Site 10, bottom right: Site 11.

Left: Site 12, right: Site 13