**Temporal and land-cover trends in freshwater fish communities in New Zealand's rivers: an analysis of data from the New Zealand Freshwater Fish Database – 1970 – 2007**

**Prepared for the Ministry for the Environment**

**Wellington**

**March 2009**

**Mike Joy**

**Ecology – Institute of Natural Resources**

**Massey University**

**Contents**

## Executive summary

1. Over the past 30 years there has been a big increase in sites entered into the New Zealand Freshwater Fish Database (NZFFD). Most of this increase has been at pasture and indigenous forest sites. In the last decade (2000 – 2007) the number of pasture sites sampled was more than the number of sites for all other land-cover classes combined.
2. Thirty-seven years of freshwater fish and crustacean presence/absence data were obtained from the NZFFD; that was all entries on flowing water dating from January 1970 to June 2007 and consisted of 22,546 sites.
3. To enable between site comparisons an Index of Biotic Integrity (IBI) was used as it takes into account natural elevational and distance from coast variation in fish communities caused by the largely migratory New Zealand fish fauna.
4. Clear differences in IBI scores were found in relation to land cover. Sites in native vegetation catchments had significantly higher scores and more species than sites in pasture and urban catchments, while those in tussock land cover had the lowest scores.
5. Analysis of IBI scores over time revealed a significant reduction in average IBI scores for the past 37 years, especially over the last decade.
6. Investigation of the temporal trends by land-cover type showed the biggest declines were at pasture, tussock, and urban sites, while exotic forest sites showed no significant change and there was a significant improvement at native forest and scrub sites.
7. Where data was available for the same reaches sampled repeatedly over time these were analysed for changes. These showed more declines than improvements, although the differences were small. This study identified a shortage of repeatedly sampled sites.
8. The measures used in this study are only based on presence/absence data, thus the results are inherently conservative because fish species will show reduced abundance long before they become locally extinct.
9. This analysis highlighted the necessity for a set of long term repeatedly sampled monitoring sites for the whole country and the need to have a consistent sampling protocol.

## Introduction

The assessment of the state of fresh waters in New Zealand and worldwide has been dominated by one-off chemical and field metered snapshots. Analysing flowing water using this approach has clear limitations as these measures vary seasonally, daily and even hourly and moreover they are all flow and rainfall influenced. This inherent variability means that long-term trends are missed due to the difficulty of finding statistically significant trends from the natural variability in such data. While these snapshots may be useful for later diagnosis they are not suitable for assessment of ecosystem condition. One solution to this problem is to use the biota (biomonitoring or bioassessment) because the resident life forms integrate all possible snapshot "water quality" measures over meaningful time scales. This realisation has changed the approach taken by many developed countries as epitomised by the European Unions Water framework directive[1].

One aspect of the flowing freshwater biota particularly suited to biomonitoring is fish; they are long lived, relatively easily captured and are highly valued by the community. At the heart of bioassessment worldwide[1] is the comparison between the biota that would be at a site in the absence of human impact with what is found there now. This can be achieved in a number of ways; either through predictive models that allow for site specific predictions and comparisons or less directly through an index that has multiple metrics and is calibrated to local conditions.

When making comparisons between sites the number of species can only be used if there has been repeat sampling at that reach over time. There has been a clear consensus that distribution of New Zealand freshwater fish fauna is driven largely by elevation and distance from the coast due to the prevalence of migratory fish species (McDowall 1998, Joy et al. 2000, Joy and Death 2001). This results in the number of species decreasing with increasing distance from the sea and elevation.

To enable between-site comparisons it is necessary to use a method that takes this into account. A Fish Index of Biotic Integrity (F-IBI) can achieve this requirement as fish community expectations are based on elevation, distance and existing conditions.

Using fish communities to define freshwater ecosystem integrity has had a relatively long history. The Index of Biotic integrity using fish was originally developed in the USA by James Karr during the early 1980s and is now used worldwide (Ganasan and Hughes 1998, Roth et al. 1998, Toham and Teugels 1999, Drake and

---

[1] In Europe, the European Water Framework Directive (WFD2000/60/EC; European Commission, 2000) introduces the obligation for its member states to achieve and maintain good ecological status for all water bodies. Such ecological status must be assessed as a deviation from the reference condition, measuring the Ecological Quality Ratio (EQR = Observed/Expected) for different quality elements (macroinvertebrates, diatoms, macrophytes and fish). While some European countries have adopted the multimetric approach in assessing biological communities (e.g. (Hering et al. 2006)), others have chosen to develop predictive systems based on the multivariate approach.

Pereira 2002). A F-IBI has been developed for all New Zealand (Joy and Death 2004) and regionally for the Waikato (Joy 2006, 2007), Auckland (Joy 2004a), Hawke's Bay (Joy 2005) and Wellington (Joy 2004b). The index of biotic integrity allows for the assessment of the fish community at a site that is independent of the elevation or distance from the sea. These two parameters are critical as the majority of the fish fauna in New Zealand is migratory. The IBI score is based on the sum of 12 scores for different aspects of fish communities; the maximum value is 60 and the minimum is 0 if there are no native fish present.

In the last year the fish IBI has been further modified for the Auckland and Waikato Regional Councils with the inclusion of a recent statistical development; quantile regression (Chen 2005). This change resulted in more accurate scoring of sites but also meant that IBI scores were higher for many sites than previous IBI indices. This is because the previous process of fitting lines by eye overestimated the slopes of lines as the densities of data points could not be accurately ascertained. The new quantile IBI approach was used in the analysis for this report.

The aim of this report is to identify national trends in freshwater fish communities over the past four decades, as an indicator of ecosystem condition, and identify any relationships between trends in fish communities and land use.

## Methods

### *Data sources*

The fish data came from the New Zealand Freshwater Fish Database (NZFFD) maintained by the National Institute of Water and Atmospheric Research (NIWA) and contains records of fish distribution for around 100 years (McDowall 1991). The data has been supplied by many individuals and institutions and in 2008 contained more than 23,000 records. Each entry includes the site location details and the species of fish and large crustaceans found there, using a number of survey methods. The amount of detail varies from only presence/absence and no habitat details, to complete site descriptions and detailed abundance and fish size measures. However, due to the differences in survey methods and measures of abundance all data were converted to presence/absence, as comparison requires consistent levels of data accuracy. To analyse the trends in the database related to land use the River Environment Classification (REC)[2] land-cover classes were applied to all sites. The major land-cover classes; pastoral, urban, indigenous forest, scrub, and exotic forest were used.

[2] http://www.niwa.cri.nz/ncwr/rec

### *Statistical analyses*

The relationship between land-cover and fish communities were analysed by comparing the mean IBI scores using an analysis of variance Proc ANOVA (SAS 2000). ANOVA compares the means between two or more samples; the result is an F–value, which is the test statistic and a P-value, which is the statistical significance of any differences. Temporal trends in fish community structure were analysed by comparing IBI scores over years and decades for all sites and then by individual REC classes to find the land-cover types underlying the differences and trends. Statistical analysis of temporal trends was done using general linear regression models PROC GLM (SAS 2000). To visualise these trends, mean decadal scores and variances were plotted. General linear models incorporate a number of statistical models including ANOVA and, when there is just one dependant variable, as in this case, then they can also be referred to as a multiple regression. As with the ANOVA, the F value is the test statistic and the P value is the measure of the statistical significance.

The reaches that had multiple sampling events were assessed for changes in IBI scores over time using a Spearman Rank Correlation (PROC CORR (SAS 2000)). The results were reported as the number of significant positive or negative relationships. To further investigate the changes at sites sampled more than once, sites that had been sampled before 2000 – 2007 were compared with the 2000 – 2007 period by counting the number of reaches that had more or less species.

### *Data limitations and interpretation of IBI scores*

The fish distribution data used in this report were not collected expressly for this analysis, rather they are a collection of sites sampled for many reasons by many different operators. Thus, there will be differing levels of sampling intensity and ability of operators; notwithstanding this the large number of sites should override this limitation to some extent. Furthermore, it is likely that sampling efficiency has improved over the 40 years, but the effect of this would be a tendency to increase scores over time. To help get around the shortcomings mentioned above of variable sampling intensity and ability, only presence/absence (p/a) data were used, and abundance data where available was converted to p/a data. However, the limitation of using p/a data is that there is a tendency to underestimate changes to fish communities because the reduction in a species abundance happens long before local extinction.

The use of the land-cover classes for sites that may have changed land-cover over time is a limiting factor that is difficult to quantify. However, sites are less likely to go from pasture into other land-cover classes e.g., scrub or indigenous forest, but the reverse is more likely, and given the recent intensification of farming the change from scrub or indigenous forest land cover into pastoral land cover is possible.

The IBI score can range between 0 for no fish and 60. The IBI was developed using data from the NZFFD for the period 1980 to 2002. The scoring process involves the top 33% of the sites that score the maximum within each of the six IBI metrics. This ensures that the expectations and scoring are reasonable for the area it is developed for (Joy & Death 2004). For New Zealand, 5% of the sites scored the maximum of 60. This is the level recommended by Karr (1981) with the upper species richness line in the original development of the IBI in America. For this national analysis the emphasis was on changes over time and within land-use types rather than what the scores mean in relation to condition, as this is better done at a regional or catchment scale. However, for clarity a higher IBI score indicates higher quality.

## Results

Thirty-seven years of freshwater fish and crustacean presence/absence data were obtained from the NZFFD. That was all entries on flowing water dating from January 1970 to June 2007. This consisted of 22,546 sites, over a broad geographic coverage (Fig. 1).

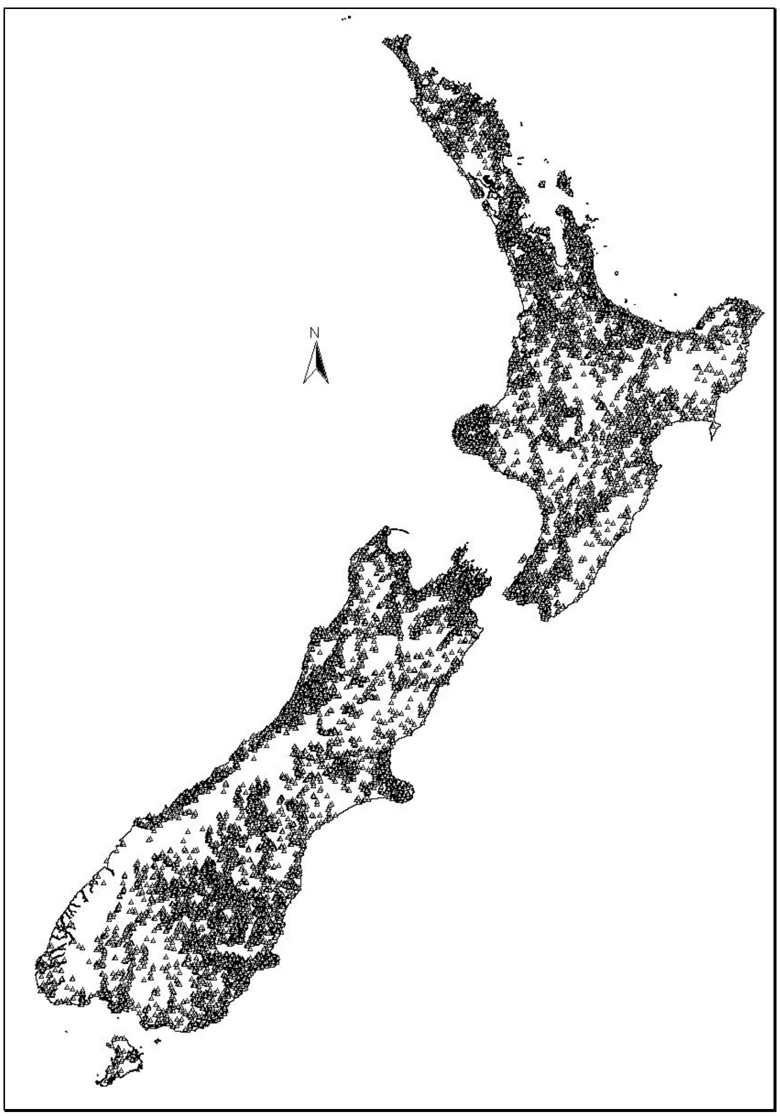


**Figure 1** Locations of the 22,546 sites in the NZFFD from 1970 – 2007

### *Sampling trends*

The number of sites added to the database has increased over time but there were different patterns of increase related to land cover. The number of sites sampled in exotic forest, urban and scrub land cover showed a gradual increase, approximately doubling every decade. Sampling indigenous forest sites increased at a much greater rate but peaked in the 1990s, while pasture sites increased exponentially over the entire period. Between 2000 and 2007 the number of pasture sites sampled was more than all the other classes combined (Fig. 2).

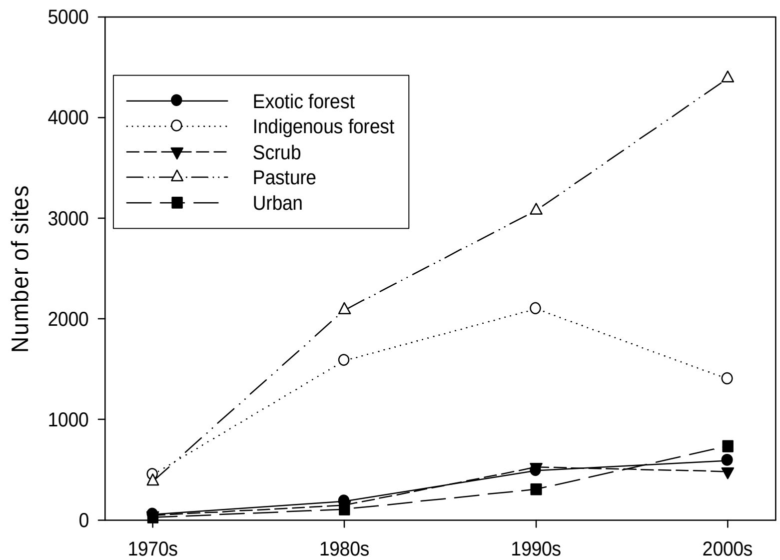


**Figure 2** Sites added to the database (NZFFD) over decades by land-cover type (Note: 2000s includes 2000 to 2007)

### *Index of biotic integrity scores in relation to land use*

The average IBI score was significantly higher at indigenous forest and scrub sites than the other land cover classes, and tussock was significantly lower than all other land-cover classes (ANOVA $F_{7,22538} = 247$; $P < 0.0001$) (Table 1; Fig. 3). Pasture sites had the next lowest scores but were not significantly different from urban, exotic and unvegetated (bare land) sites.

**Table 1** Descriptive statistics for IBI scores by River Environment Classification (REC) class

| *REC class* | *Pasture* | *Urban* | *Exotic forest* | *Bare land* | *Tussock* | *Indigenous forest* | *Scrub* |
|---|---|---|---|---|---|---|---|
| Mean | 29.68 | 30.33 | 30.47 | 31.4 | 17.98 | 36.22 | 36.51 |
| Median | 32 | 34 | 32 | 40 | 0 | 38 | 40 |
| Standard deviation | 17.62 | 16.69 | 19.93 | 20.09 | 19.89 | 17.85 | 19.27 |
| Standard error | 0.18 | 0.49 | 0.55 | 0.88 | 0.37 | 0.24 | 0.56 |
| Number of sites | 9,932 | 1,167 | 1,319 | 522 | 2,806 | 5,530 | 1,194 |

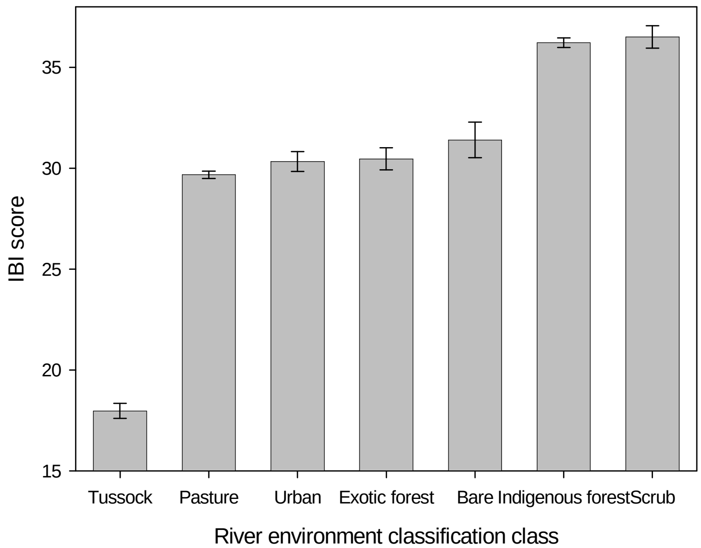


**Figure 3** Average IBI score for all sites grouped by River Environment Classification (REC) land-cover class (whiskers = Standard Error).

***Temporal trends in biotic integrity scores 1970 - 2007***

The IBI scores for all sites show there has been a significant decline in indigenous freshwater biodiversity over the past 37 years. This decline was significant for both years and decades (Table 2) with the biggest reduction in the last decade (Fig. 4). To assess which of the land-cover classes contributed to this decline the different classes were analysed separately. Indigenous forest sites showed a significant increase for both years and decades, peaking in the 1990s (Fig. 5; Table 2). Pasture sites showed a significant decrease in IBI scores for both years and decades especially in the last decade (Fig. 6; Table 2). The sites in scrub land cover showed no significant trend over decades but did for years (Fig. 7; Table 2). Urban sites showed a significant decline in IBI scores over the 37 years for both years and decades (Fig. 8; Table 2). The exotic forest sites showed a dip in the 1990s but there was no significant linear trend for both years and decades (Fig. 9; Table 2). Tussock sites showed a significant decline in IBI scores for both years and decades (Fig 10; Table 2).

**Table 2** Results of regression analyses for all sites and land cover classes using IBI scores for years and decades. Note: trend is significant if *P-value* is less than 0.05 (ns = not significant).

| *REC land-use class* | *Direction of change* | *Number of sites* | *All years* | | *Decades* | |
|---|---|---|---|---|---|---|
| | | | *F-value* | *P-value* | *F-value* | *P-value* |
| All sites | negative | 22545 | 191.2 | 0.0001 | 223.7 | 0.0001 |
| Pasture | negative | 9931 | 92.0 | 0.0001 | 118.4 | 0.0001 |
| Tussock | negative | 2805 | 21.1 | 0.0001 | 38.83 | 0.0001 |
| Indigenous | positive | 5529 | 41.5 | 0.0001 | 24.7 | 0.0001 |
| Urban | negative | 1157 | 29.6 | 0.0001 | 19.9 | 0.001 |
| Scrub | negative/ns | 1193 | 3.9 | 0.047 | 1.21 | 0.27 |
| Exotic | ns | 1318 | 2.4 | 0.13 | 0.09 | 0.77 |

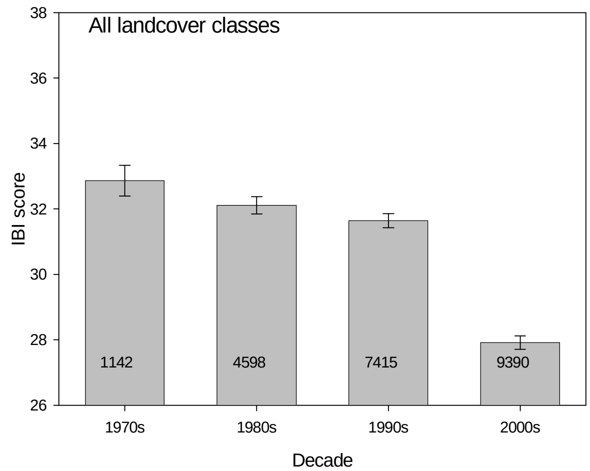


**Figure 4** Average decadal IBI score for all sites (number of sites inside bars, whiskers = Standard Error)

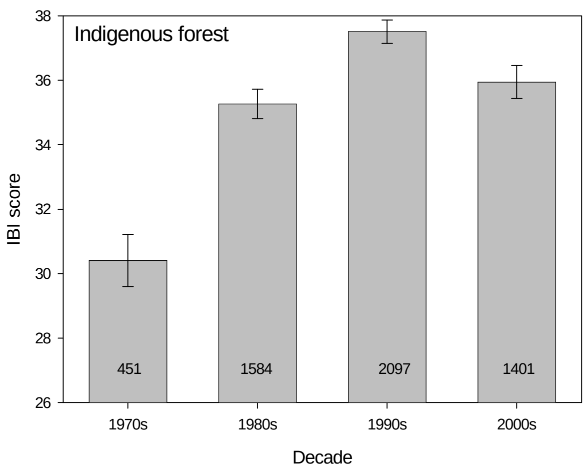


**Figure 5** Average decadal IBI scores for River Environment Classification (REC) land-cover Indigenous forest sites (number of sites inside bars, whiskers = Standard Error).

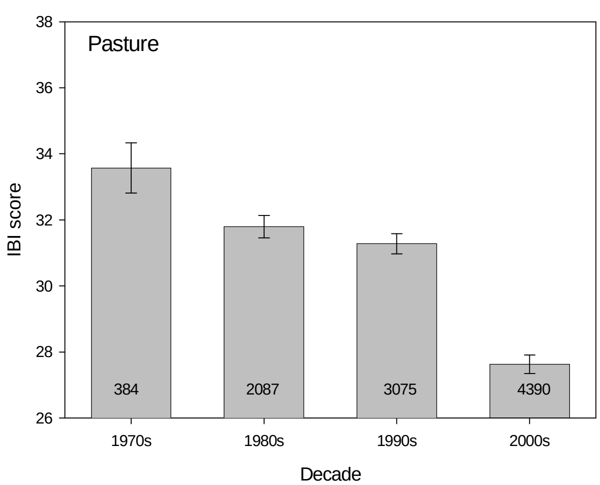


**Figure 6** Average decadal IBI scores for River Environment Classification (REC) land-cover pasture sites (number of sites inside bars, whiskers = Standard Error)

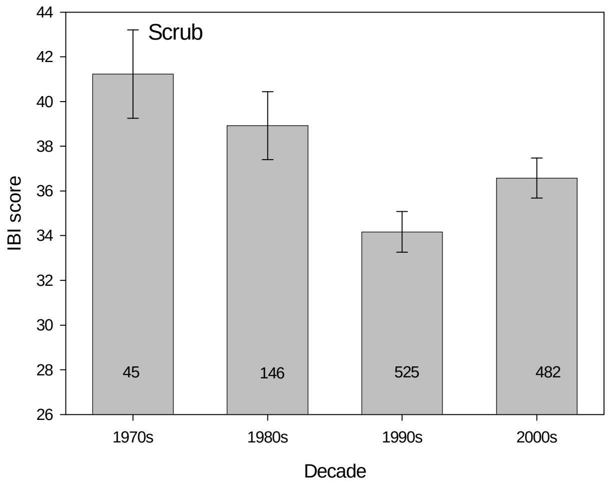


**Figure 7** Average decadal IBI scores for River Environment Classification (REC) land-cover scrub sites (number of sites inside bars, whiskers = Standard Error)

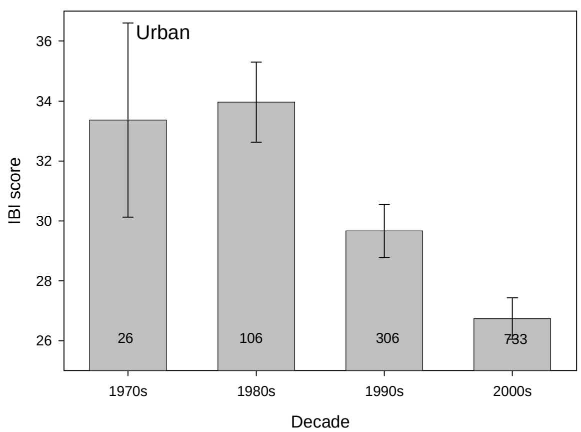


**Figure 8** Average decadal IBI scores for River Environment Classification (REC) land-cover urban sites (number of sites inside bars, whiskers = Standard Error)

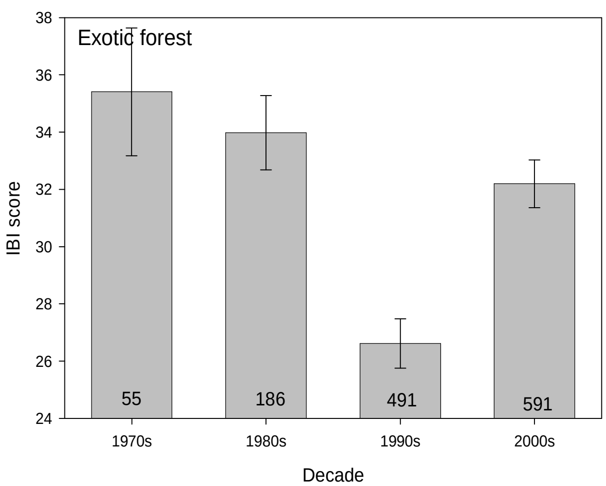


**Figure 9** Average decadal IBI scores for REC land-cover exotic forest sites (number of sites inside bars, whiskers = Standard Error)

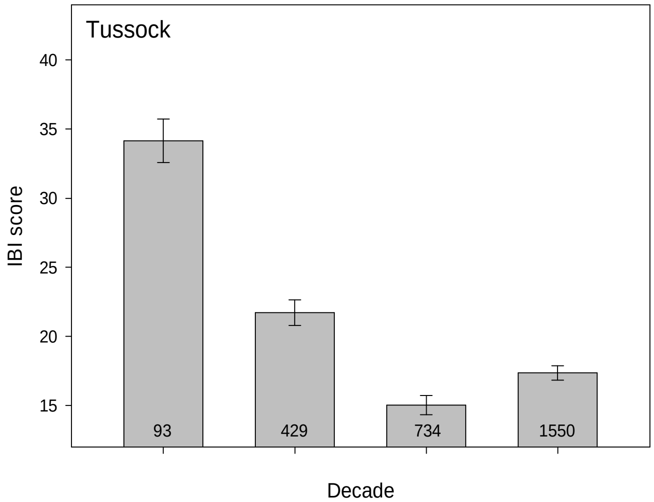


**Figure 10** Average decadal IBI scores for River Environment Classification (REC) land-cover tussock sites (number of sites inside bars, whiskers = Standard Error)

### *Changes in number of species at resampled reaches*

A number of reaches in the database had been sampled more than once over the 37 years and these were analysed to look for changes. This could be done using the number of species as they were directly comparable, which negated the need to use the IBI. The number of sites that had more or fewer species in 2000 – 2007 compared to the 5-year periods between 1970 and 1999 were counted. There were consistently more sites sampled on the same reach that had less species than more species in the 2000s, when compared to the 5-year periods between 1975 and 1999 (Fig. 11). The only exception was the 1970 – 1975 period when more sites gained species than lost species over time. However, there were only 40 sites that were sampled in 1970 – 1975 that changed from the 2000s.

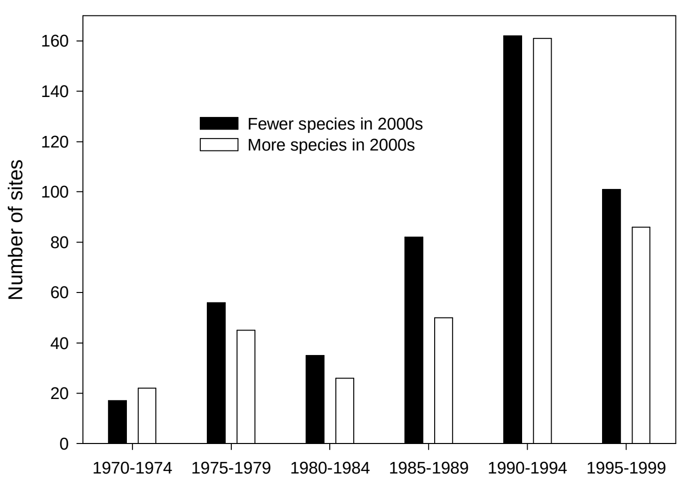


**Figure 11** Comparison between the numbers of species at reaches sampled before and then again in 2000s

Using rank correlation analysis there were very few significant differences found in IBI scores over time for the stream reaches that had multiple sampling events. However, there were more significant positive than negative changes for native vegetation, no differences at exotic forest, but more negative correlations at pasture and urban sites (Fig. 12).

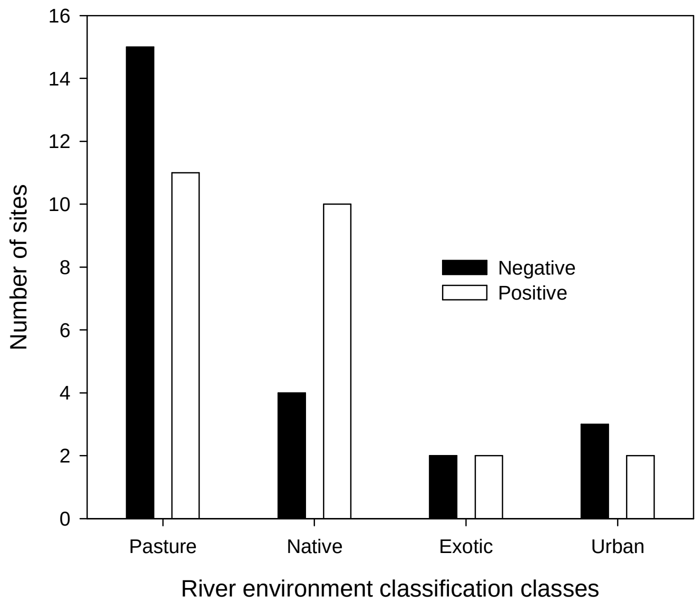


**Figure 12** Significant Spearman rank correlations for IBI scores over time for stream reaches with multiple sampling events by River Environment Classification land cover

**Discussion**

This analysis of New Zealand fish community data at 22,500 river and stream sites showed a significant decline in fish IBI scores nationally over the past 37 years. This represents a significant decline in indigenous freshwater biodiversity in these waterways. This decline indicates that freshwater ecosystem condition has also declined nationally over this time, particularly over the last decade.

Strong relationships between fish biotic integrity scores and land-cover type were revealed using the River Environment Classifications (Fig. 3). The term biotic integrity is based on the concept that to function, an ecosystem must have all its component parts, thus any loss of parts is effectively lost integrity (Jackson and Davis 1994, Karr 1995, Yoder and Rankin 1998, Barbour et al. 2000). Consequently, the difference shown in IBI scores within different land uses reveals the link between integrity and ecosystem function. Fish IBI scores were significantly higher at indigenous/scrub vegetation sites, and lower at urban, pasture and exotic forest sites. This strong association between fish IBI and land use shows the influence degradation of terrestrial systems has on freshwater ecosystems.

The very low scores for tussock land cover are likely to be the result of the fact that these sites are generally a long way from the sea and are at high elevation. The majority of native fish are migratory, thus these sites generally have low diversity or cannot be reached by fish. This is supported by the low mean and median scores (median = 0, mean = 17.98; Table 1) and the fact that 1453 or 51% of the tussock sites had no fish species present, so scored an IBI score of 0.

The low scores for exotic forest were not expected as the general consensus is that exotic forest is not as big an impact on streams as pasture or urban land use. While growing, the forests do provide protection for streams. However, the clear felling harvest system commonly used in New Zealand forests means that post-harvest impacts are extensive. In Figure 9 there is an obvious dip in IBI scores for exotic forest sites in the 1990s. This is likely to reflect the large amount of harvesting that took place in New Zealand, following an intense period of planting in the 1960s.

Indigenous forest sites revealed a significant increasing trend with a peak in the 1990s (Fig. 5). This could be related to improving conditions in the lower reaches of rivers with removal of or improvements in point source discharges, allowing increased fish access to the headwaters. This increase in scores at indigenous forest sites could also be related to improvements in sampling efficacy over time, the focussing of sampling efforts into more remote areas and/or an increase in the use of spotlighting as a survey tool over time. There may also have been a trend over time towards less or more intensive sampling or change in area sampled. However, any of these changes should not be specific to any one land-use class so do not unduly influence the results presented here.

Because of a lack of consistent detail in the database on sampling intensity and fish abundance, all data used in this analysis were necessarily reduced to presence/absence. This restriction means that all results are inherently conservative. This is because any species within a fish community/population will show a gradual decline before local extirpation even with relatively sudden environmental impacts. Thus, for a reduction in IBI score, fish species must become extinct at that reach. Accordingly, the observed changes exposed in this report reveal the endpoints of longterm cumulative changes to fish communities.

To take advantage of fish as a bioassessment resource that can track change in ecosystem status nationally over time, this analysis has highlighted the need to have a national set of regularly sampled sites. To keep the valuable history already in existence, a set of sites at reaches with a record of quality sampling events in the past should be added to systematically. This could be done regionally and combined into a national long-term monitoring dataset.

This initial analysis of the NZFFDB has revealed that, while there is considerable information available, any future analysis would be much improved by consistently sampled long term single site data as well as further investigation into the different sampling methods. Furthermore, analysis of the trends in relation to biological parameters such as the migratory status of species and whether or not they are part of a recreational or commercial fishery could add important information. The conservation status of New Zealand freshwater fish species is being undertaken at present by the Department of Conservation and this process could be improved by targeted species specific analysis of the NZFFDB.

**Acknowledgements**

Thanks to Jody Richardson at NIWA for extracting data from NZFFDB and to Nicola Atkinson and Fiona Death for help with data manipulation. Thanks also to Russell Death and Bruno David for discussions about analyses and interpretation. Thanks to Graham McBride and Bruno David for comments on the final report.

**References**


Barbour, M. T, Swietlik, W. F, Jackson, S. K, Courtemanch, D. L, Davies, S. P, and Yoder, C. O. 2000. Measuring the attainment of biological integrity in the USA: a critical element of ecological integrity. Hydrobiologia **422**:453-464.

Chen, C. 2005. An introduction to Quantile regression and the QUANTREG procedure. *in* Proceedings of the Thirtieth Annual SAS Users Group International Conference. SAS Institute Inc., Cary, NC.

Drake, M. T, and Pereira, D. L. 2002. Development of a fish-based index of biotic integrity for small inland lakes in central Minnesota. North American Journal of Fisheries Management **22**:1105-1123.

Ganasan, V, and Hughes, R. M. 1998. Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh), India. Freshwater Biology **40**:367-383.

Hering, D, Feld, C. K, Moog, O, and Ofenbock, T. 2006. Cook book for the development of a Multimetric Index for biological condition of aquatic ecosystems: Experiences from the European AQEM and STAR projects and related initiatives. Hydrobiologia **566**:311-324.

Jackson, S, and Davis, W. 1994. Meeting the goal of biological integrity in water-resource programs in the US Environmental Protection Agency. Journal of the North American Benthological Society **13**:592-597.

Joy, M. K. 2004a. A Fish Index of Biotic Integrity (IBI) for the Auckland Region. Bioassessment software in, Palmerston North. Ecology Group and Centre for Freshwater Ecosystem Modelling and Management. Massey University Palmerston North.

Joy, M. K. 2004b. A fish Index of Biotic Integrity (IBI) for the Wellington Region. Ecology Group and Centre for Freshwater Ecosystem Modelling and Management. Massey University Palmerston North

Joy, M. K. 2005. Point-click-fish a predictive model of fish occurrence for the Hawkes Bay region. Ecology Group and Centre for Freshwater Ecosystem Modelling and Management. Massey University Palmerston North.

Joy, M. K. 2006. A Predictive Model of Fish Distribution and Index of Biotic Integrity (IBI) for Wadeable Streams in the Waikato Region. Environment Waikato Technical Report 2006/07 Ecology Group and Centre for Freshwater Ecosystem Modelling and Management. Massey University Palmerston North.

Joy, M. K. 2007. A new fish Index of Biotic Integrity using Quantile regressions: the Fish QIBI for the Waikato Region. Ecology Group and Centre for Freshwater Ecosystem Modelling and Management. Massey University Palmerston North.

Joy, M. K, and Death, R. G. 2001. Control of freshwater fish and crayfish community structure in Taranaki, New Zealand: dams, diadromy or habitat structure? Freshwater Biology **46**:417-429.

Joy, M. K, and Death, R. G. 2004. Application of the index of biotic integrity methodology to New Zealand freshwater fish communities. Environmental Management **34**:415-428.

Joy, M. K, Henderson, I. M. and Death, R. G. 2000. Diadromy and longitudinal patterns of upstream penetration of freshwater fish in Taranaki, New Zealand. New Zealand Journal of Marine and Freshwater Research **34**:531-543.

Karr, J. R. 1981. Assessments of biotic integrity using fish communities. Fisheries **6**:21-27.

Karr, J. R. 1995. Ecological integrity and ecological health are not the same. Pages 97-109 *in* P. Schultz, editor. Engineering Within Ecological Constraints. National Academic Press, Washington.

McDowall, R. M. 1991. Freshwater fisheries research in New Zealand: processes, projects, and people. New Zealand Journal of Marine and Freshwater Research **25**:393-413.

McDowall, R. M. 1998a. Driven by diadromy: its role in the historical and ecological biogeography of the New Zealand freshwater fish fauna. Italian Journal of Zoology **65**:73-85.

McDowall, R. M. 1998b. Use of freshwater fishes as indicators of environmental health - further analyses. A report prepared for the Ministry for the Environment MFE70509, National Institute of Water and Atmospheric Research, Christchurch.

Roth, N, Southerland, M., Chaillou, J, Klauda, R, Kazyak, P, Stranko, S, Weisberg, S, Hall, L, and Morgan, R . 1998. Maryland biological stream survey: Development of a fish Index of Biotic Integrity. Environmental Monitoring and Assessment **51**:89-106.

SAS. 2000. SAS/STAT Version 8.2. SAS Institute Inc, Cary, North Carolina.

Toham, A. K, and Teugels, G. G. 1999. First data on an index of biotic integrity (IBI) based on fish assemblages for the assessment of the impact of deforestation in a tropical West African river systems. Hydrobiologia **397**:29-38.

Yoder, C. O, and Rankin, E. T. 1998. The role of biological indicators in a state water quality management process. Environmental Monitoring and Assessment **51**:61-88.