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Can juvenile yellowbelly and sand flounder abundance indices and environmental variables predict adult abundance in the Manukau and Mahurangi Harbours?

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EXECUTIVE SUMMARY

McKenzie, J.R.; Parsons, D.M.; Bian, R. (2013).
Can juvenile yellowbelly and sand flounder abundance indices and environmental variables predict adult abundance in the Manukau and Mahurangi Harbours?

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Annual surveys of juvenile flatfish year class strength could potentially offer a cost-effective and fishery independent method of quantifying adult flatfish abundance. We investigated the utility of existing juvenile survey data from the Manukau and Mahurangi Harbours. The majority of our analysis was performed on yellowbelly flounder data from the Manukau Harbour, as catches there are from a discrete area and are almost entirely composed of yellowbelly flounder.

We initially used the juvenile survey data to predict adult abundance while varying the level of unknown recruitment before and after the time series. Results from this simulation suggested that the seven year time series should be capable of providing reasonable estimates of adult yellowbelly flounder biomass for a five to seven year period. When the estimates of adult yellowbelly flounder abundance generated from the simulation were compared to the commercial set net CPUE index, however, only a weak correlation was evident. A longer time series of juvenile flatfish abundance would be required to fully assess the utility of juvenile surveys.

We also assessed the influence of environmental variables on flounder abundance. Consistent correlations were obtained for a variety of environmental variables for juvenile sand and yellowbelly flounder in the Manukau, but not Mahurangi Harbour. The influence of environmental variables on adult YBF catch in the Manukau Harbour was even more evident. These correlations suggested that decreasing oxygen and increasing ammonia and turbidity may have negatively affected yellowbelly flounder recruitment success. When these results are considered alongside the declining trends in flatfish abundance in FLA1, estuarine water quality may be a significant factor affecting the sustainability of the flatfish fishery. Our overall recommendation is therefore that a broader study of trends in water quality, at least for the west coast North Island harbours, be undertaken and potential water quality mitigation measures investigated.

1. INTRODUCTION

While nine flatfish species are combined under the FLA management code, the majority (97%) of the flatfish catch taken within the FLA1 Quota Management Area (upper half of the North Island; Figure 1) comprises two species: the yellowbelly flounder (*Rhombosolea leporina*, YBF) and the sand flounder (*R. plebeia*, SFL). Juveniles of both species are found in high abundance within estuarine systems, but as adults SFL are predominantly caught in sandy bays out to 40 m depth (and dominate flatfish catches within the Hauraki Gulf and Firth of Thames). Adult YBF, however, remain within harbour systems and dominate catches in harbours such as the Manukau and Kaipara. In addition to being comprised of two species, the FLA1 stock is probably made up of a number of different sub-stocks (different harbours and coastal areas within the FLA1 QMA show different trends in abundance (Kendrick & Bentley 2011)). Both YBF and SFL are fast growing and short lived species, generally only surviving to three to four years of age and rarely to five or six years of age (Colman 1974, Perks 1985). The combination of short lifespan and fast growth implies that localised populations may be relatively resistant to over fishing, and may experience fluctuations in abundance driven by variation in annual recruitment. If local flounder abundance is largely determined by recruitment variation it should be possible to predict adult abundance by monitoring annual recruitment (also known as Year Class Strength (YCS)) via netting surveys within harbours, or even by monitoring strongly correlated environmental variables such as sea surface temperature (e.g. Coburn & Beentjes 2005). Such an approach is untested, but has the potential to provide a relatively low cost and fishery independent prediction of flounder abundance.

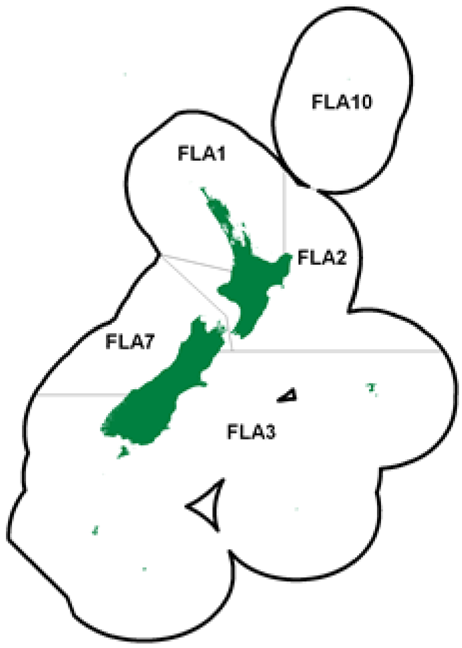


Figure 1: Flatfish Quota Management Area boundaries.

For two locations within the FLA1 stock, time series of juvenile (0^+) flatfish densities (number of individuals per square metre from beach seine net surveys already exist (M. Morrison, NIWA, Auckland, unpublished data; Francis et al. 2005; Figure 2; Figure 3). These data provide an opportunity to assess whether juvenile flatfish abundance indices are useful in predicting subsequent adult catch. The two NIWA juvenile flounder time series are for the Manukau (seven year time series, 2001–2007) and Mahurangi Harbours (six year time series, 2002–2007). The objectives of this study were to: (1) assess the utility of juvenile flounder YCS indices (suitably lagged) in predicting patterns seen in adult commercial catch per unit effort (CPUE), (2) investigate whether environmental variables were correlated with this juvenile YCS index, and (3) investigate whether annual fluctuation in environmental variables (suitably lagged) correlate with patterns seen in flounder commercial CPUE.

a.

b.

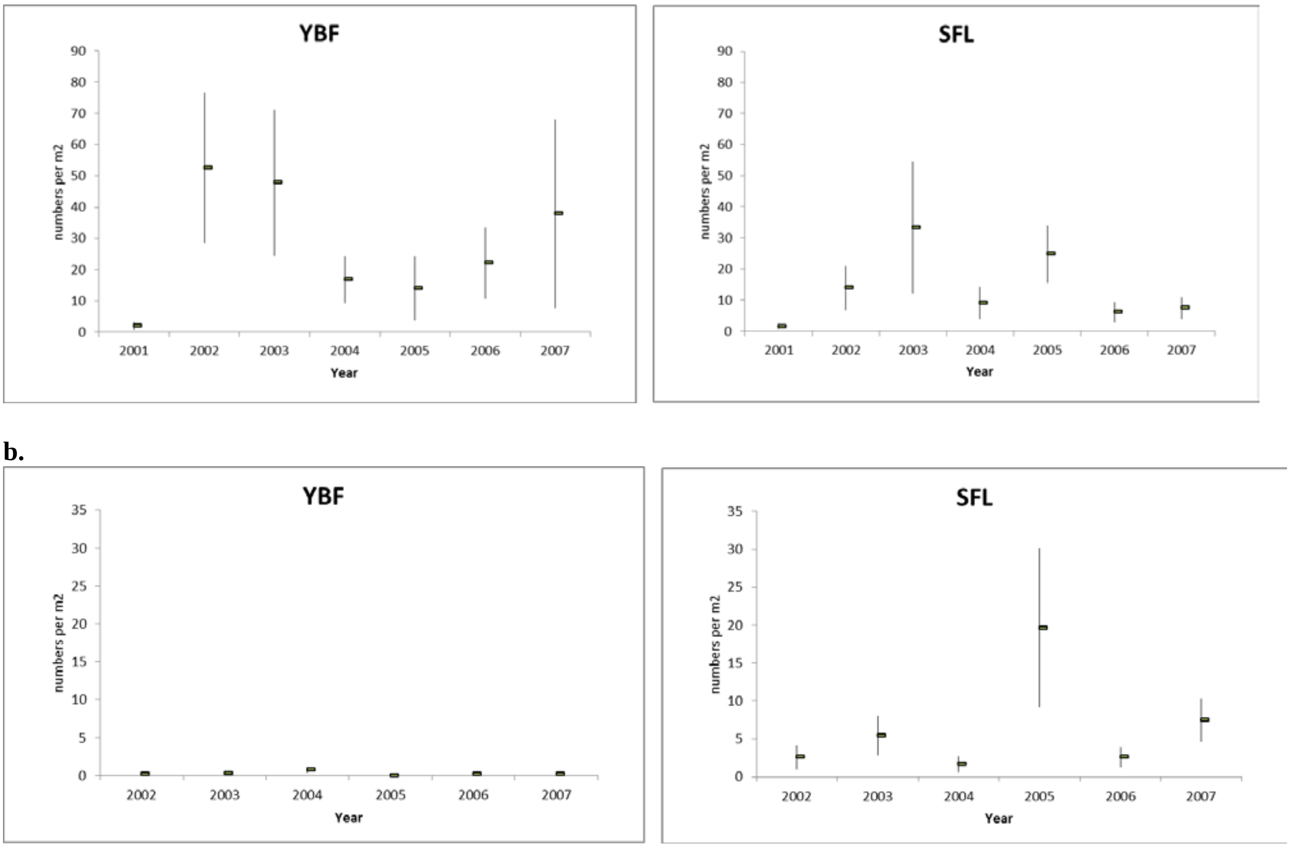


Figure 2: Juvenile flounder abundance estimates (YCS) derived from the Manukau and Mahurangi Harbour netting surveys (a. Manukau; b. Mahurangi) all sampling took place during February (M. Morrison, NIWA, Auckland, unpublished data).

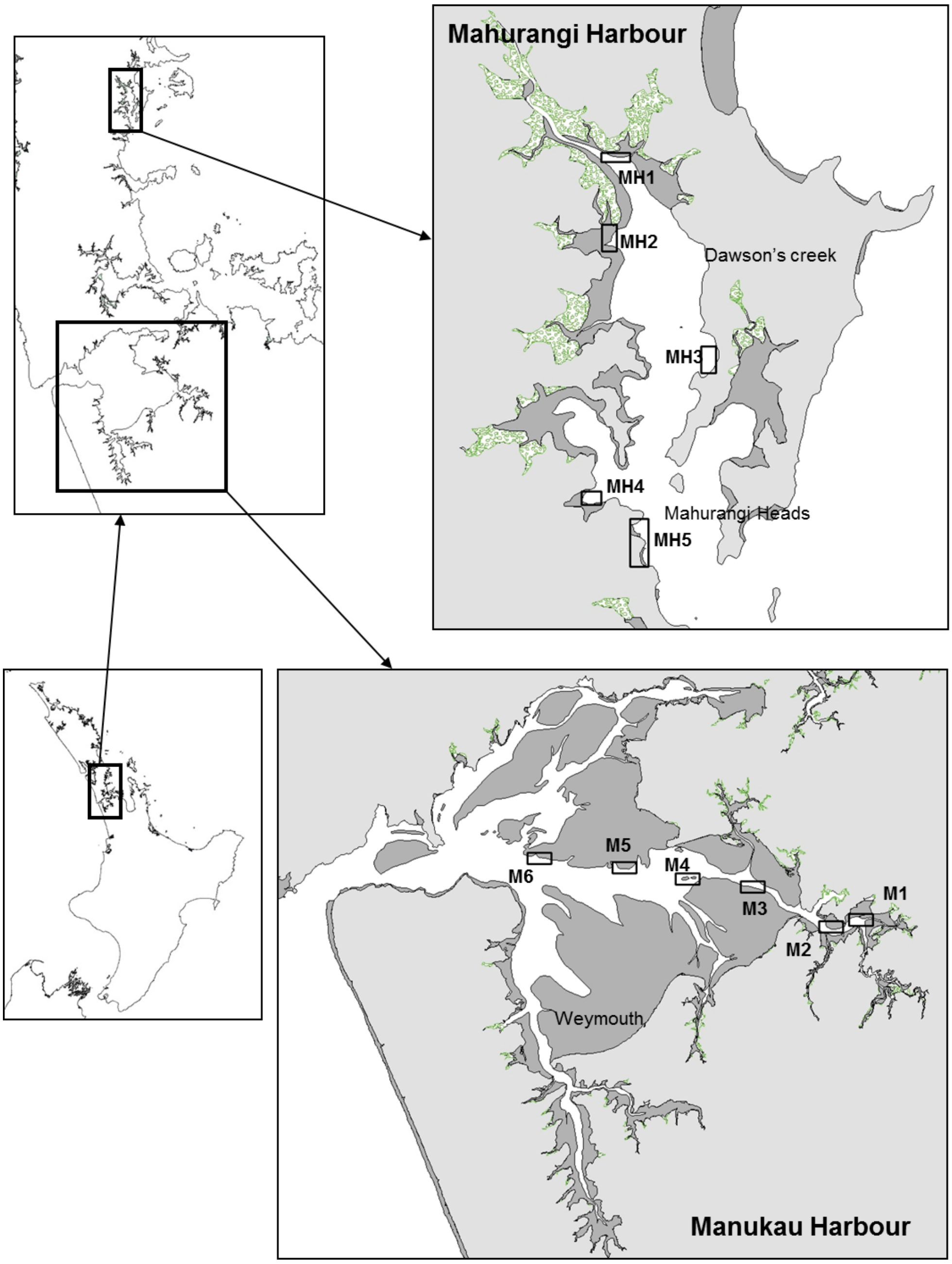


Figure 3: Manukau and Mahurangi Harbours juvenile flounder NIWA sampling sites 2001–2007 (numbered boxes).

2. METHODS

2.1 Correlation between environmental variables and the juvenile YCS index

It is likely that environmental factors determine the YCS within the Manukau and Mahurangi Harbours. Therefore, we sought to determine which environmental variables were important by correlating a selection of likely effects with the juvenile recruitment index obtained from the Manukau and Mahurangi Harbour netting surveys (Figure 2). Auckland Council routinely measures various water quality variables at both Weymouth in the Manukau Harbour, and within the Mahurangi Harbour at Dawsons Creek and at the Mahurangi Heads (Figure 3; Appendix 1). In addition we obtained measurements of local rainfall at each site as well as the Trenberth index (Z1: pressure differential between Auckland and Christchurch) and the Southern Oscillation Index (SOI; pressure differential between Darwin and Tahiti).

Where environmental variables did not conform to normal distributions they were logged. For both SFL and YBF in the Mahurangi, and YBF in the Manukau, the juvenile index was then correlated with monthly averages for each environmental variable with successive lagging back to eight months prior to the date when juvenile fish were sampled (February).

Prior to correlating the environmental and YCS data, the environmental data were assessed for cross correlations (collinearity). An iterative process was followed where one of each pair of highly correlated variables (those pairs where the absolute value of the Pearson statistic was greater than 0.6) was dropped. The process continued until a set of variables was obtained in which none of the cross-correlations exceeded the 0.6 threshold.

2.2 Derivation of adult flounder abundance estimates from Manukau harbour YCS data using a stock assessment model

Species-specific catch and effort data was used to investigate the utility of juvenile YCS surveys in predicting adult flatfish abundance. Catch and effort information for flounder in the Ministry for Primary Industries (MPI) databases are not separated by species for the Hauraki Gulf, nor can catches specific to the Mahurangi Harbour sub-area be identified. However, MPI records do contain flounder catch and effort data specific to the Manukau Harbour and although the specific flounder species are not identified in the data, it is reasonable to assume that catches are almost entirely composed of YBF (Kendrick and Bentley 2011). We therefore confined our investigation to YBF abundance within the Manukau Harbour.

An additional consideration was that flounder catch at any point in time will be composed of multiple year classes. Therefore, to accurately assess the utility of the juvenile YCS index, the relative influence of different year classes within the catch must be known. To address this concern an age-structured stock assessment model[1] was used to generate estimates of YBF spawning stock biomass (SSB) in the Manukau Harbour. Model inputs were: the Morrison netting survey YCS time series for YBF (NIWA unpublished data); YBF von Bertalanffy growth parameters specific to the Manukau Harbour (Perks 1985), a total mortality estimate based on age data derived through projecting the 1985 Perks age data through commercial length frequency data collected in 1987(unpublished NIWA data); a YBF length-weight relationship (Mutoro 1999); commercial catch was fixed at 250 tonnes per year an amount consistent with recent annual catches (Kendrick and Bentley 2011).

The influence of recruitment from years before and after the period where juvenile surveys were conducted was an additional concern. To investigate this, we ran the model over an extended time period at different assumed levels of recruitment for the years before and after the period of YCS observations. When the different model runs converged with each other our interpretation was that the model had adequate predictive power (i.e. the influence of unknown recruitment variability was minimal) for those periods. The predicted relative adult abundances generated were then compared with the Catch Per Unit Effort (CPUE) index generated from commercial catch data (described in section 2.3).

[1] The model was constructed using NIWA's CASAL (C++ Algorithmic Stock Assessment Laboratory) stock modelling building software (Bull et al. 2010).

2.3 CPUE analysis

2.3.1 Manukau Harbour YBF CPUE abundance index comparison with model predictions of adult abundance

Commercial set net catch information specific to the Manukau Harbour (where catches are almost entirely YBF) was available between the 1989–90 and 2009–10 fishing years. It was therefore possible to derive a CPUE index of abundance for YBF specific to the Manukau Harbour using these data.

Before a CPUE analysis was performed a fleet of core vessels was selected on the basis that they contributed a large proportion of the overall catch and fished with consistency throughout the time series. Core vessels were selected using the methodology outlined by Kendrick and Bentley (2011). Catch data were then groomed line by line, paying particular attention to large or small entries for catch, duration of fishing, net length and mesh size. Often data errors could be identified as they were outside the range expected for a particular variable. When this occurred the variable could often be replaced with the usual value for that fisher. Where catch was in doubt, the entire record was deleted. A stepwise Generalised Linear Model (GLM) analysis was then performed on the catch and effort data where the response variable was log positive catch (i.e. zero catch events were not included). Covariates offered to the model included vessel (24 levels), month (12 levels), season (4 levels), year (21 levels), log duration, log mesh size and log of net length. The stepwise process successively added and removed parameters so that the most explanatory parameters (in terms of overall improvement in model residual deviance) were identified. The process was stopped when the addition of a term did not increase the overall residual deviance by more than 2%.

2.3.2 Importance of including environmental variables in CPUE standardisations

A second CPUE analysis was undertaken in which water quality variables collected at Weymouth by Auckland Council (as detailed in section 2.2) were also offered as covariates to the GLM. Water quality conditions in the Manukau Harbour are likely to be most influential on flounder abundance around the critical period when a year class is spawned. To test this in a stepwise GLM standardisation the various water quality sample values had to be time-lagged to correspond to the time when the adult year classes within the fishery were likely to be settling as larvae in the harbour. Age sampling data suggests that the fishery is almost entirely composed of two and three year olds in approximately equal proportion by weight (this report). YBF catch was therefore matched with the average of water quality measurements from two and three years previously for a specific month. YBF are likely to settle to the substratum during late spring (November; Park 1983), furthermore, it is likely that water conditions at or around the time of settlement influence recruitment success (YCS). To test for water quality parameters and months of influence we conducted five separate CPUE standardisations, each based on water quality in a different month (June to November). In each of these separate analyses the water quality measurements from the month of choice were applied to the relevant year that matched the year classes of adult fish within the fishery at that time (i.e. all fishing events within a particular year were associated with the same water quality values).

3. RESULTS

3.1 Correlation between environmental variables and the juvenile YCS index

After removing highly correlated environmental variables the set of environmental variables used in the analysis were (see Appendix 1 for description):
Turbidity (log);
Water temperature;
Suspended solids (log);
Nitrate (log);
Ammonia (log);
Dissolved oxygen;
Total phosphorus (log);
Dissolved Reactive Phosphorus (log);
Faecal coliform (log);
Monthly rainfall (log);
Trenberth index (Z1);
Southern Oscillation Index (SOI).

Correlation coefficients (Pearson) corresponding to different time lags between environmental variables and the abundance of juvenile SFL and YBF in the Manukau and Mahurangi Harbours are presented in Figure 4 to Figure 7. Correlation coefficients greater or less than plus or minus 0.5 were of interest; whereas values in the central range around 0 suggest a weak relationship between YCS and the environmental variable in a given month. Where a trend or cycle in correlation strength with month was present that is also of interest, whereas random perturbations were less so. While there are many correlations to consider across the 12 environmental variables, two species and within two areas, we highlight some of the more noticeable patterns below.

In the Manukau Harbour, a similar pattern of correlation was evident for both SFL and YBF abundance when compared to dissolved oxygen content (Figure 4 and Figure 5). This pattern suggested that four to seven months prior to the juvenile surveys, abundance of both species was positively correlated with dissolved oxygen content. Also in the Manukau Harbour, both SFL and YBF have a strong negative correlation with the SOI when lagged by three to seven months and both species had a strong positive correlation with turbidity between when lagged by two and three months and a strong negative correlation with turbidity when lagged between four and six months (Figure 4 and Figure 5).

In the Mahurangi Harbour the pattern of correlations with environmental variables were less consistent when compared across SFL and YBF. For example, the correlation with dissolved oxygen took a different form for SFL compared to YBF (Figure 6 and Figure 7). There was also evidence of a negative correlation with the SOI for SFL; this is similar to the pattern seen for SFL and YBF in the Manukau Harbour. Mahurangi SFL also showed similar negative monthly correlations with water temperature to the Manukau Harbour SFL and YBF trends.

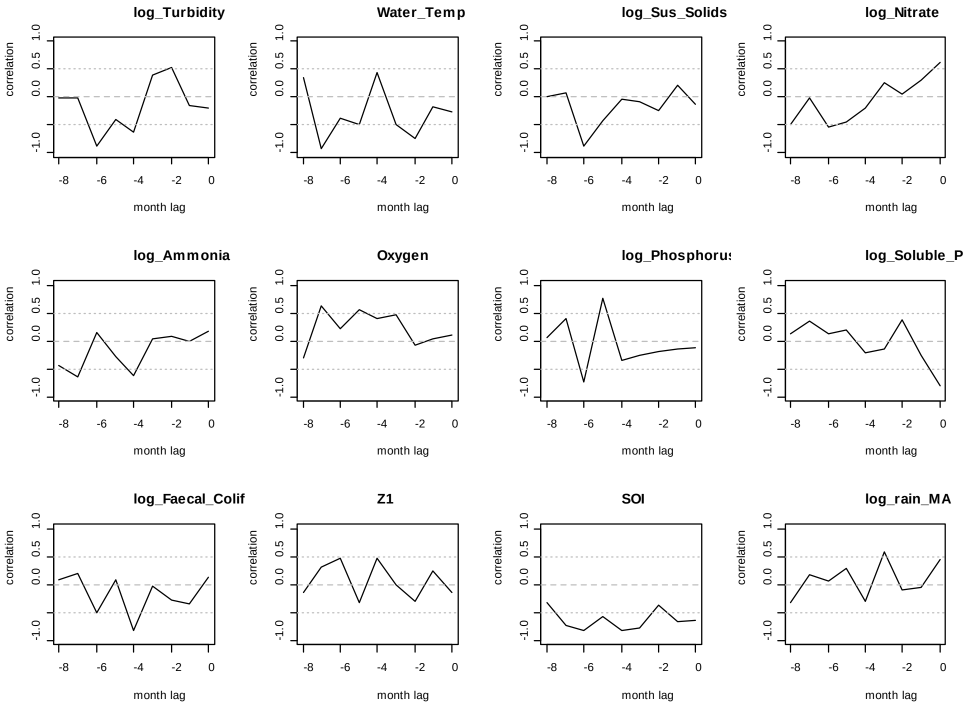


Figure 4: Pearson correlation values between juvenile SFL abundance (Manukau Harbour) and environmental variables lagged by up to eight months prior to sampling (February).

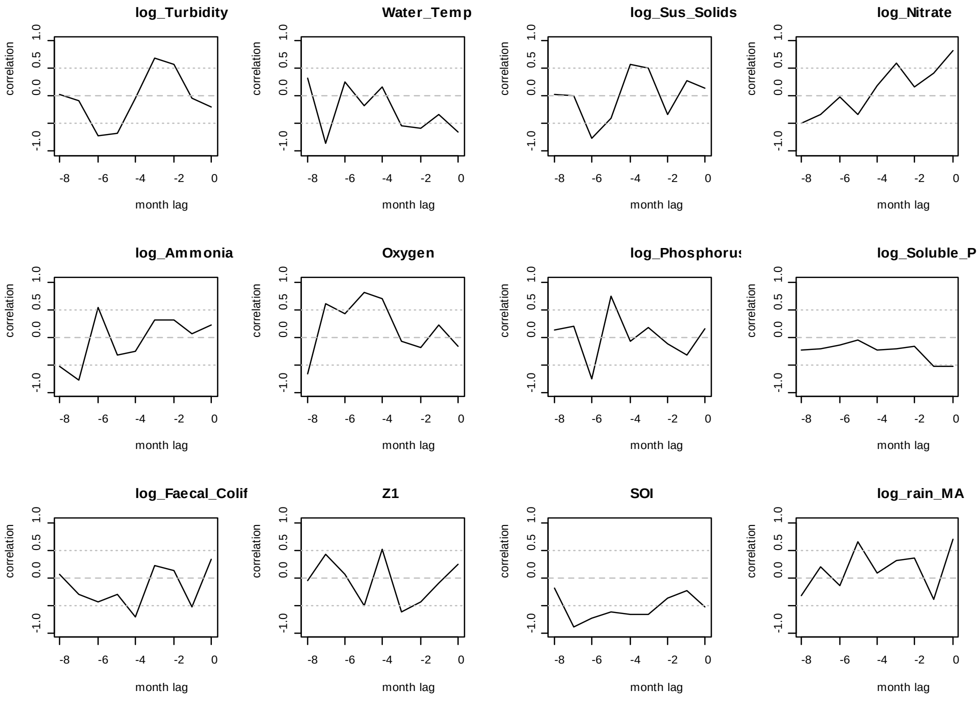


Figure 5: Pearson correlation values between juvenile YBF abundance (Manukau Harbour) and environmental variables lagged by up to eight months prior to sampling (February).

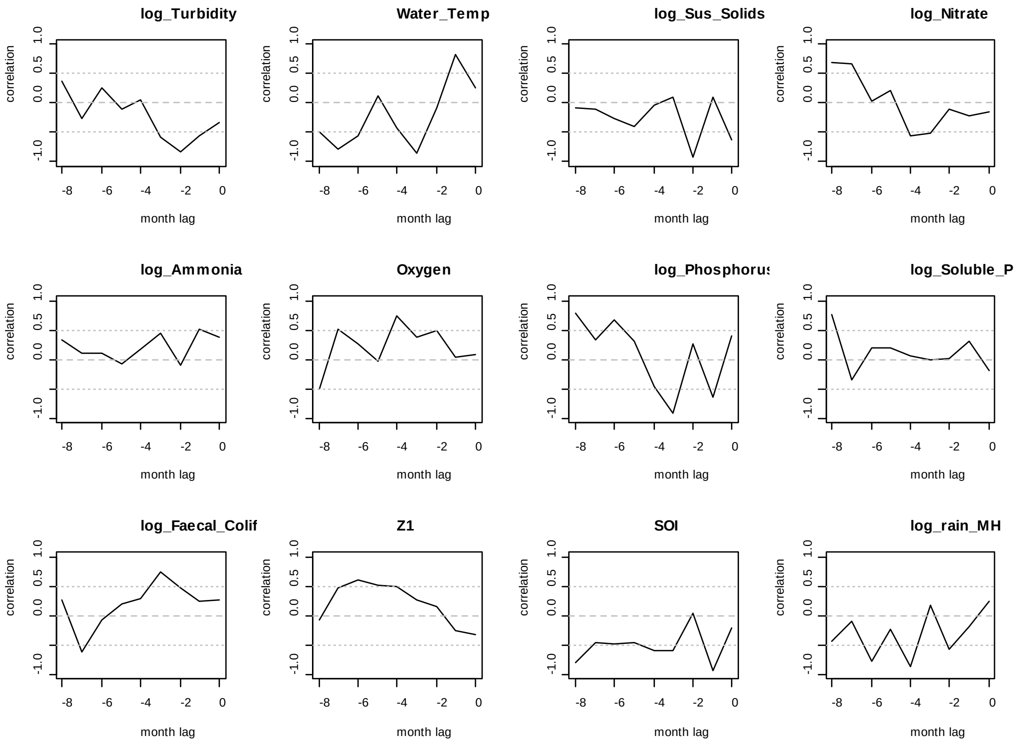


Figure 6: Pearson correlation values between juvenile SFL abundance (Mahurangi Harbour) and environmental variables lagged by up to eight months prior to sampling (February).

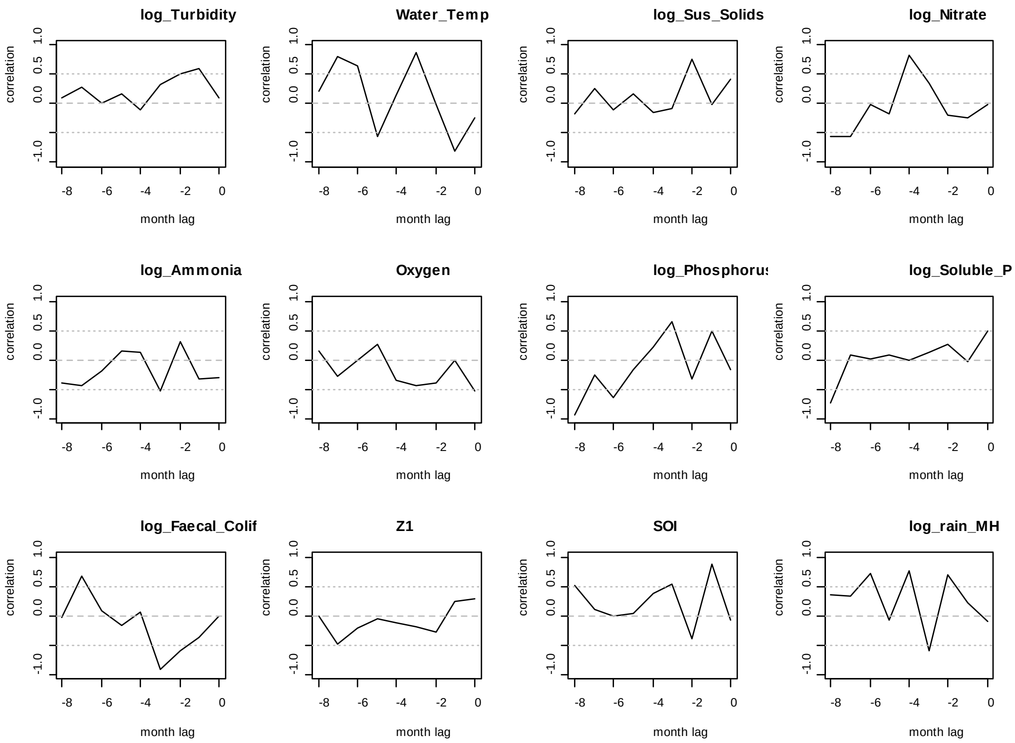


Figure 7: Pearson correlation values between juvenile YBF abundance (Mahurangi Harbour) and environmental variables lagged by up to eight months prior to sampling (February).

3.2 Utility of juvenile YCS indices in predicting adult flatfish catch

3.2.1 Manukau Harbour adult flounder abundance as derived from YCS data using a stock assessment model

3.2.1.1 Model inputs

Growth estimates
Estimates of YBF growth in the Manukau Harbour were derived from data collected in 1983–84 by Perks (1985). Von Bertalanffy growth parameter estimates (Figure 8) show distinct differences between male and female YBF.

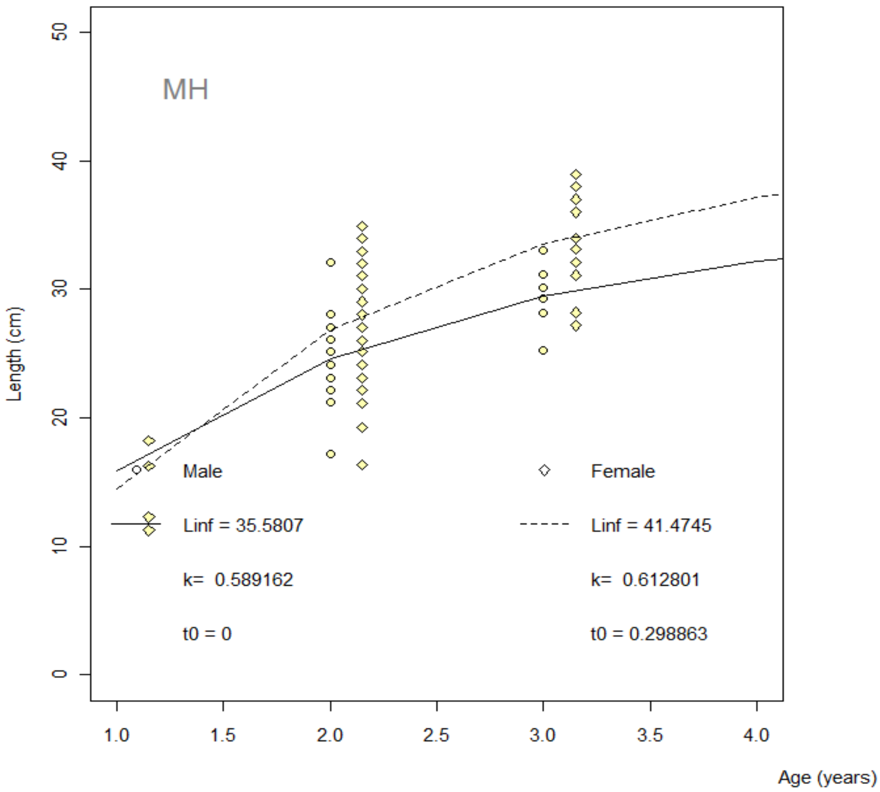


Figure 8: Von Bertalanffy growth curve fits to Manukau harbour YBF age length data collected in 1983–84 (Perks 1985).

Length weight
YBF length weight parameters were derived from data collected by Mutoro (1999) (Figure 9).

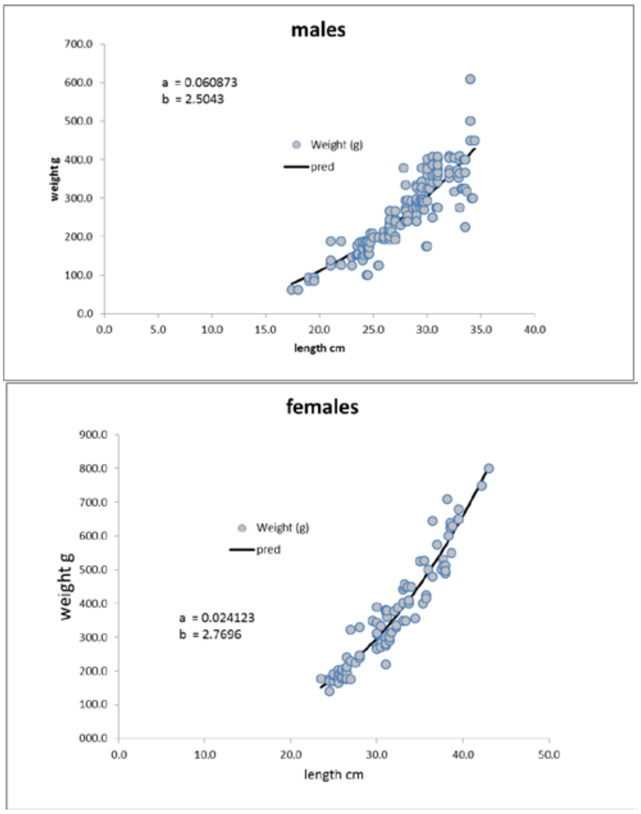


Figure 9: YBF length weight parameter estimates curves fitted to Mutoro (1999) data.

Total mortality

YBF length frequency data from the Manukau Harbour were available from commercial catch sampling undertaken in 1987 (Ministry for Primary Industries unpublished data; Appendix 2). These length data were projected through the Perks (1985) age length data as an age-length key to generate the age distribution given in Figure 10.

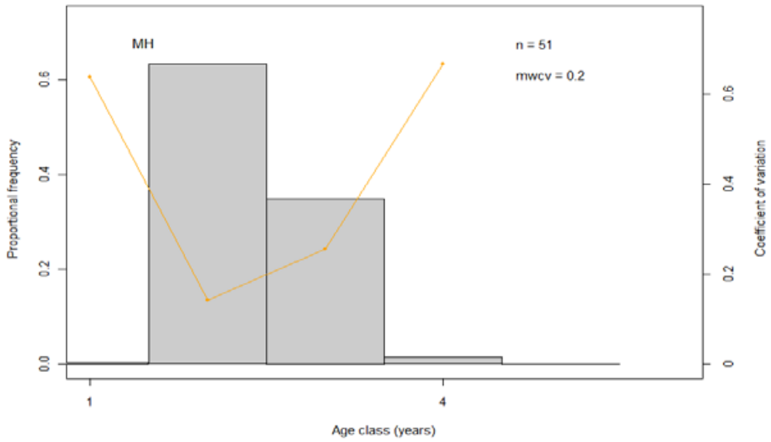


Figure 10: Age frequency of Manukau Harbour YBF produced by applying the Perks (1985) age length data to the 1987 commercial setnet length frequency. Lines show the coefficient of variation (c.v.) on the individual age estimates.

YBF catches in the Manukau Harbour setnet fishery in 1985–87 were predominantly composed of only two age classes: 2 and 3 year-olds (Figure 10); each age class contributed roughly equally to the fishery by weight. There were few 4 year-olds in the catches and no 5 year-olds; the decay rate of age cohorts equates to a total instantaneous mortality rate of 1.28.

Manukau Harbour Juvenile YCS index 2001-2007

The Morrison juvenile flounder data provided a seven-year (2001-2007) time series of relative year-class strength for Manukau harbour flounder (Figure 2).

3.2.1.2 Model relative biomass predictions based on the survey YCS estimates

The CASAL prediction of the relative adult biomass of YBF in the Manukau Harbour derived from the juvenile YCS surveys is presented in Figure 11. The contribution of “unknown” year classes to recruited stock biomass in most scenarios disappeared within two years (Figure 11), the extreme being four years as seen in the scenario where unknown recruitment was set at 10 times the mean (Figure 11). A similar converse effect is seen after the cessation of juvenile surveys (Figure 11), in that stock biomass can be predicted for a further two years. Overall the CASAL simulation results suggest that the seven year juvenile YCS survey may provide useful estimates of adult YBF biomass for a five to seven year period (Figure 11).

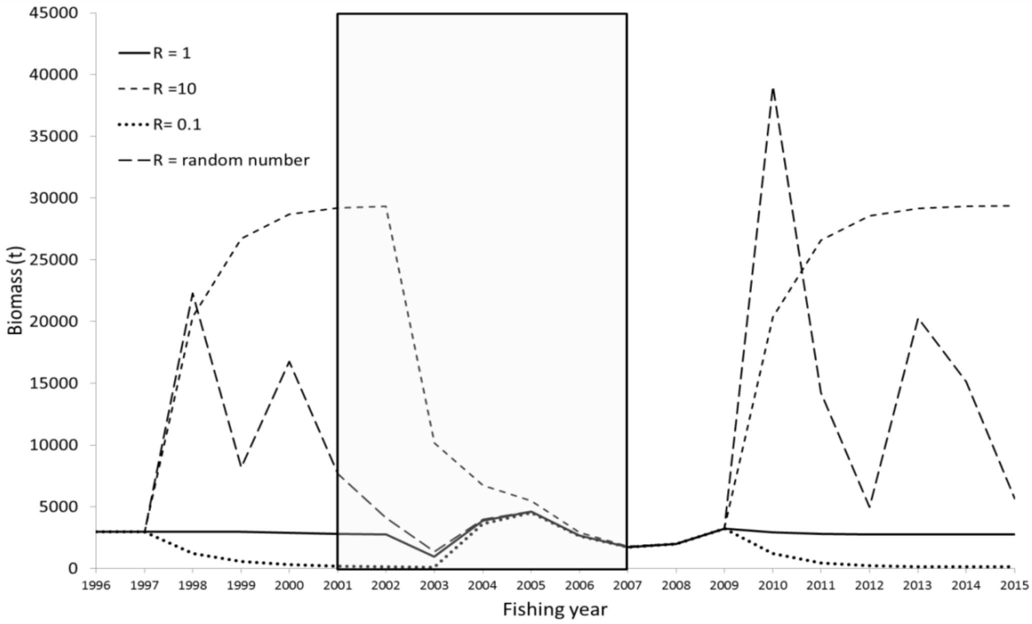


Figure 11: CASAL simulation of the relative biomass of adult YBF within the Manukau Harbour fishery. The highlighted box outlines the period when juvenile YCS surveys were conducted. The different series denote estimated biomass under different theoretical recruitment scenarios for the period before and after the YCS surveys (i.e. R = 1 implies that annual recruitment before and after the surveys was constant at 1e^{+008}; the other series simply denote a multiplier for that level of recruitment). When the different series converge with each other this indicates a time period when the CASAL simulation estimates were largely not influenced by recruitment before or after the YCS surveys.

3.2.2 Manukau Harbour commercial setnet CPUE abundance index

Covariate terms chosen in the stepwise GLM standardisation of the time-series of Manukau Harbour commercial setnet data were vessel, net-length, duration and month (Table 1).

Table 1: Variables selected by the stepwise GLM standardisation of YBF setnet catch and effort data from the Manukau Harbour. 'ns' denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded. Fishing-year (not shown in table) was forced as the first term in the model.

	Number of significant terms				
Variable	1	2	3	4	n.s.
Vessel	0.31				
Log netlength	0.14	0.35			
Log duration	0.11	0.33	0.36		
Month	0.09	0.32	0.36	**0.38**	
Season	0.09	0.32	0.36	0.38	0.38

Cyclic fluctuations in abundance, consistent with variable annual recruitment patterns, are seen in the set-net CPUE index (Figure 12). An overall downward abundance trend is evident in the CPUE series after 1991 (Figure 12), this trend is not consistent with the abundance pattern expected for a short-lived species and is unlikely to be caused by high fishing pressure alone.

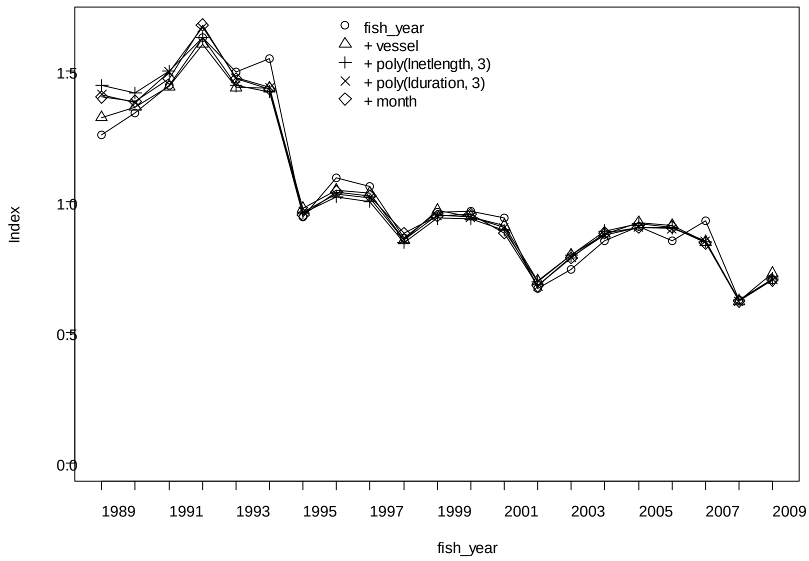
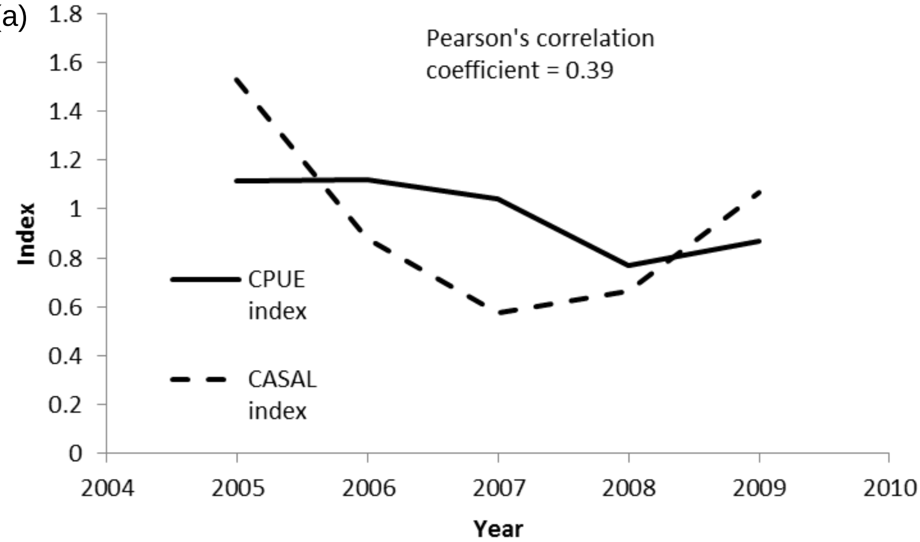


Figure 12: Step plot of GLM CPUE standardisation showing the resultant change in year index with the addition of each successive significant covariate term.

3.2.3 Correlation between adult flatfish catch and the juvenile YCS index

Figure 11 suggests we should have most confidence in the CASAL model abundance predictions for the five year period 2005 through to 2009. The CPUE and CASAL model abundance indices of YBF from the Manukau Harbour have broadly similar downward trends (Figure 13). With only five years of data in the CASAL simulation, however, the overall correlation was weak (Pearson = 0.39; linear regression R^2 = 0.15; Figure 13).

(a)



(b)

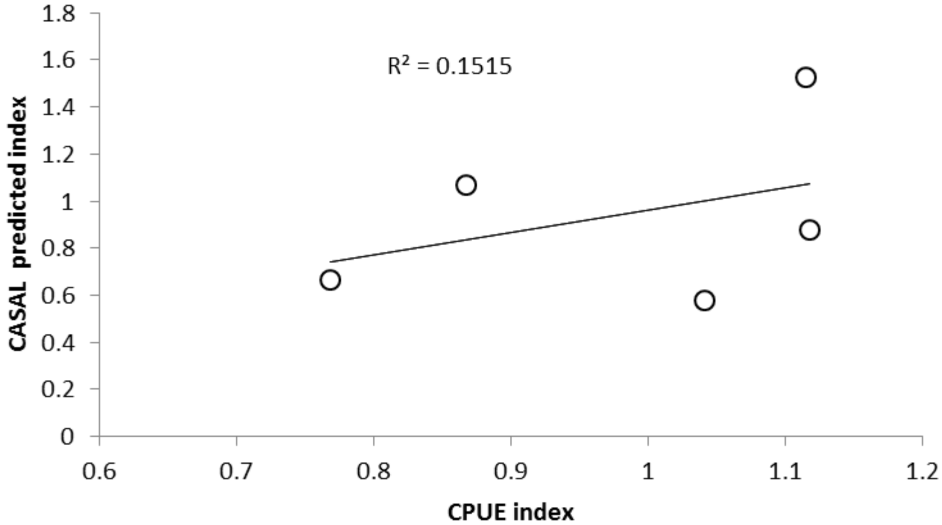


Figure 13: Abundance of YBF in the Manukau Harbour (as predicted by the CASAL model) relative to the YBF CPUE index (a), and the correlation between YBF CPUE and the CASAL prediction of YBF abundance for the Manukau Harbour.

3.3 Results from including environmental variables in Manukau Harbour setnet CPUE standardisations

None of the stepwise GLM standardisations (Section 3.1) accepted any of the lagged environmental covariates. One possible explanation is that these variables were acting as a proxy for fishing-year (i.e. were strongly correlated with the fishing-year index). To provide more insight as to the influence of environmental variables we chose to repeat the analyses dropping fishing-year from the standardisations; which would allow an environmental variable to proxy for the fishing-year effect (abundance).

When environmental variables corresponding to the month of November (average of two successive years) were offered to the GLM “ammonia concentration” was selected as the third variable in the GLM model (Table 2). The November average “ammonia concentration” showed a reasonable negative correlation (Pearson -0.66) with the base YBF Manukau Harbour CPUE index (Figure 12) as shown in Figure 14a.

Table 2: GLM output table excluding fishing year and including November average environmental variables. ‘ns’ denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded.

	Number of significant terms				
Variable	1	2	3	4	ns
vessel	0.268				
Log netlength	0.042	0.313			
Log Ammonia	0.033	0.283	0.327		
Log duration	0.032	0.284	0.322	**0.337**	
month	0.019	0.276	0.322	0.335	0.347

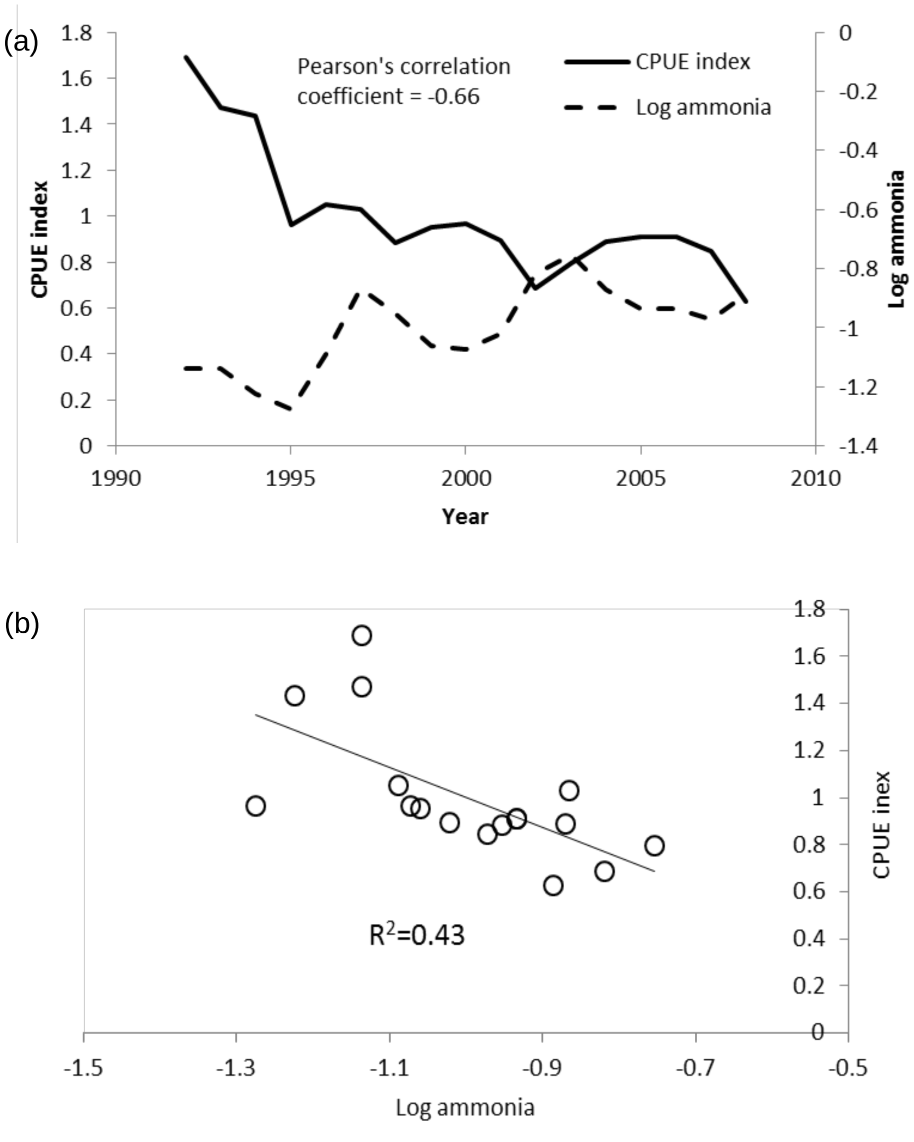


Figure 14: GLM index of Ammonia concentration in November and the YBF CPUE base index for the Manukau Harbour (a) and corresponding linear regression (b).

When environmental variables corresponding to the month of October (average of two successive years) were offered to the GLM, no environmental variables were selected by the GLM model (Table 3).

Table 3: GLM output table excluding fishing year and including October average environmental variables. 'ns' denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded.

	Number of significant terms		
Variable	1	2	n.s.
Vessel	0.266		
Log netlength	0.042	0.311	
Month	0.019	0.275	0.3211

When environmental variables corresponding to the month of September (average of two successive years) were offered to the GLM "oxygen concentration" was selected as the third variable in the GLM model (Table 4). The September average "oxygen concentration" showed a reasonable positive correlation (Pearson 0.65) with the base YBF Manukau Harbour CPUE index (Figure 12) as shown in Figure 15a.

Table 4: GLM output table excluding fishing year and including September average environmental variables. 'ns' denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded.

				Number of significant terms		
Variable	1	2	3	4	5	ns
vessel	0.268					
Log netlength	0.042	0.313				
Oxygen	0.028	0.280	0.324			
Log duration	0.034	0.283	0.320	0.335		
month	0.019	0.275	0.321	0.333	**0.346**	
season	0.014	0.273	0.319	0.331	0.344	0.346

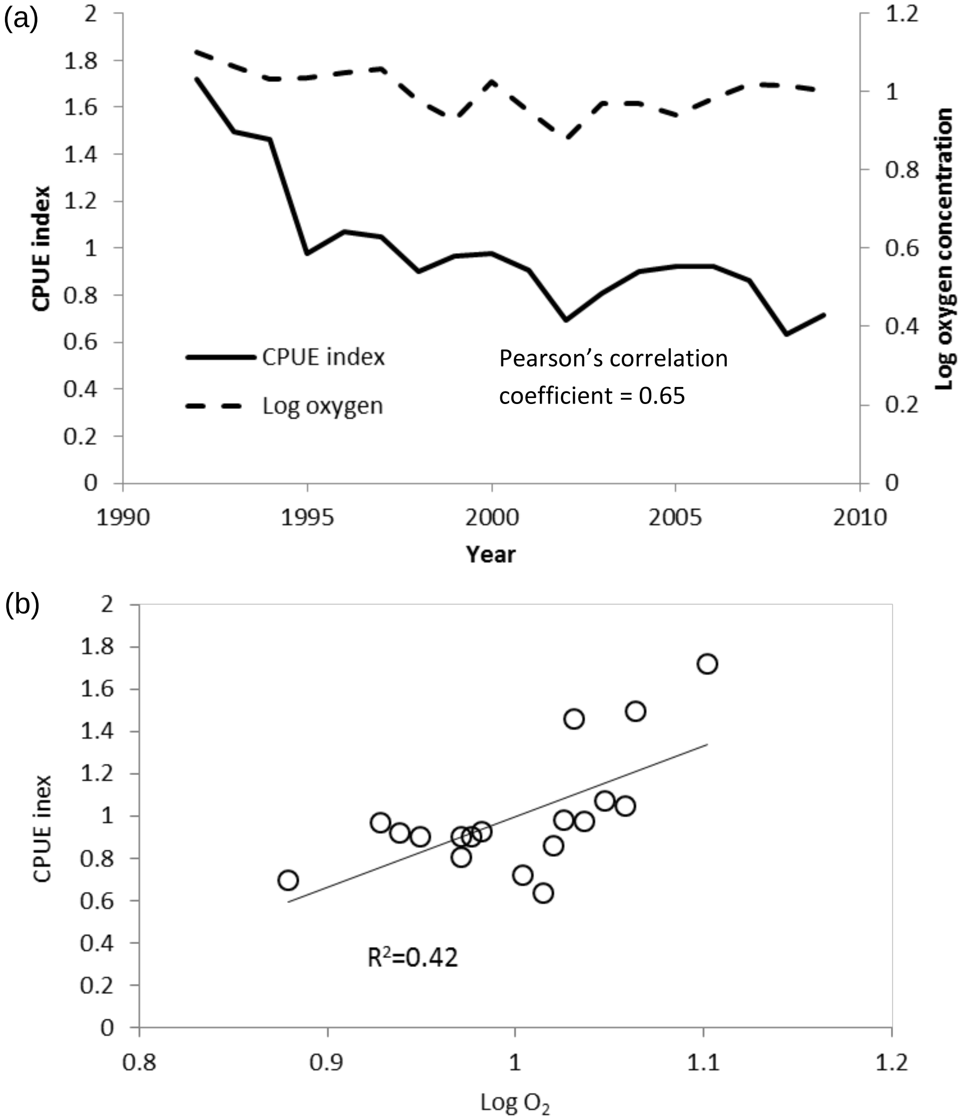


Figure 15: GLM index of Oxygen concentration in September and the YBF CPUE base index for the Manukau Harbour (a) and corresponding linear regression (b).

When environmental variables corresponding to the month of August (average of two successive years) were offered to the GLM, no environmental variables were selected by the GLM model (Table 5).

Table 5: GLM output table excluding fishing year and including August average environmental variables. 'ns' denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded.

	Number of significant terms		
Variable	1	2	n.s.
Vessel	0.266		
Log netlength	0.042	0.311	
Month	0.019	0.275	0.3211

When environmental variables corresponding to the month of July (average of two successive years) were offered to the GLM**,** turbidity was selected as the third variable in the GLM model (). The September average turbidity showed a reasonable negative correlation (Pearson -0.78) with the base YBF Manukau Harbour CPUE index (Figure 12) as shown in Figure 16a.

Table 6: GLM output table excluding fishing year and including July average environmental variables. 'ns' denotes the first non-significant term. Table shows improvement in total model R^2, final selected variable value bolded.

	Number of significant terms			
Variable	1	2	3	ns
vessel	0.268			
Log netlength	0.042	0.313		
Log turbidity	0.025	0.284	**0.329**	
Log duration	0.034	0.283	0.320	0.339

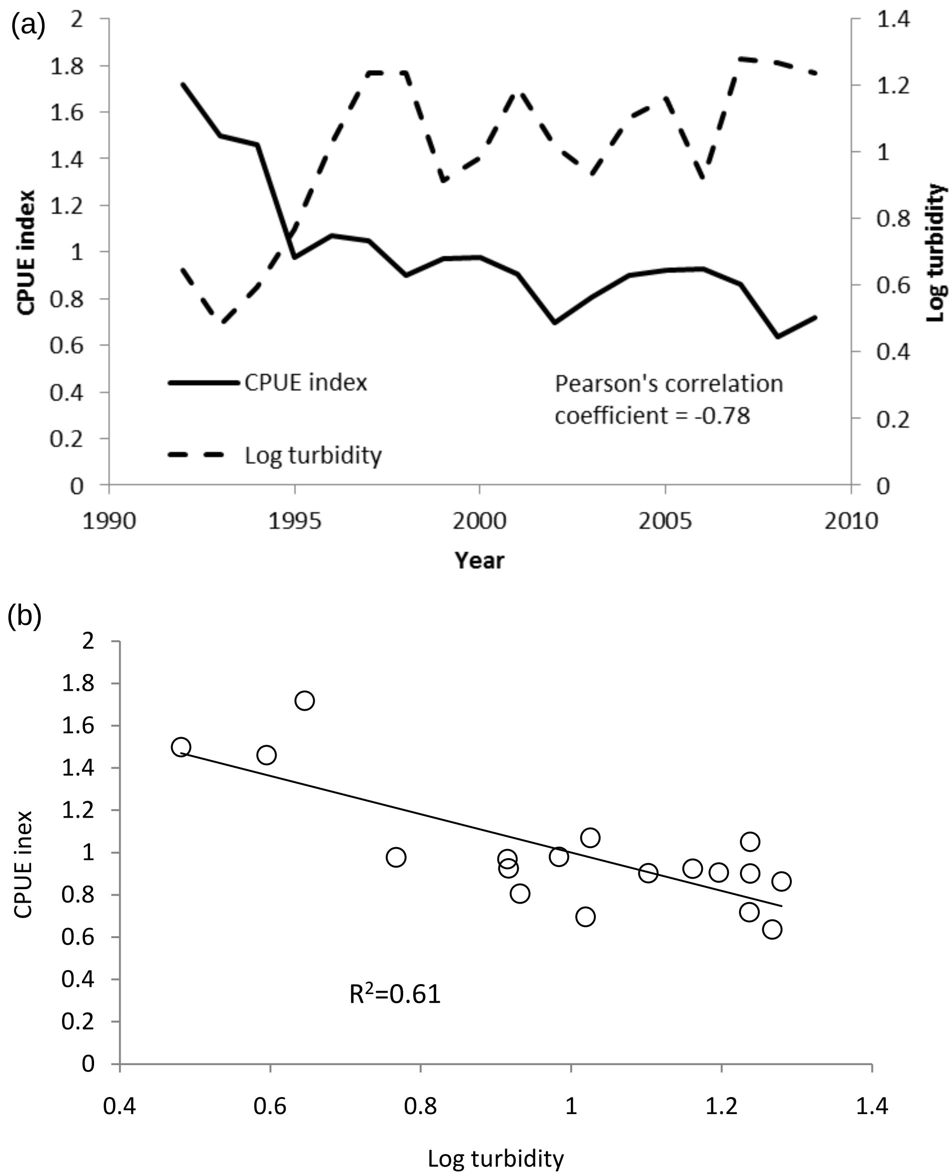


Figure 16: GLM index of "Water turbidity" in August and the YBF CPUE base index for the Manukau Harbour (a) and corresponding linear regression (b).

4. DISCUSSION

Juvenile flatfish YCS indices likely have some power, as demonstrated by the CASAL simulation, as predictors of adult relative biomass. With a juvenile YCS index of seven consecutive years, reasonable estimates of adult YBF relative biomass should be attainable for a five to seven year period. However a five year adult index from the CASAL model as derived from the NIWA Manukau harbour netting survey data was only weakly correlated with a comparable YBF CPUE index (Pearson's correlation coefficient = 0.39). The period was likely too short to conduct any meaningful comparison using a simple correlation approach. Also, the Pearson correlation did not take into account uncertainty around the respective indices; in the event a strong Pearson correlation had been seen it would not necessarily have meant the two methods were tracking the same abundance trend. What is needed is a sufficiently long time series of YCS and CPUE data to cover a distinct trend or pattern in harbour flatfish abundance. As an example; we would have needed YCS survey data covering the full 1990 decadal decline in CPUE (Figure 12), to have confidence that YCS netting surveys were tracking adult YBF harbour abundance, i.e. a viable alternative approach to CPUE. In addition, there is a possibility that changes in YBF CPUE over time may not necessarily be driven by changes in fish abundance (e.g. see Rose & Kulka (1999), Dunn et al. (2000), Maunder & Punt (2004), for a discussion of these issues). Another potential source of uncertainty is that the specific sites surveyed for juvenile YBF each year may not be representative of the sub-stock being assessed. In conclusion, a longer juvenile YCS data series would be required to fully establish the predictive power of juvenile surveys. While the initial cost of such a time series may be high, it may result in a cost-effective and fishery-independent method of assessing YBF abundance trends in the future.

In terms of the influence of environmental variables on FLA YCS, a range of correlations at different time lags were obtained for juvenile YBF and SFL in both the Manukau and Mahurangi Harbours. In the Manukau Harbour the correlation patterns of SFL and YBF were similar for dissolved oxygen concentration, the SOI and turbidity. Similarity in these correlation trends was not evident for the Mahurangi Harbour, which may have been due to the low density of YBF captured there (mean less than 0.8 YBF m^{-2}).

The series of stepwise GLM regressions on Manukau Harbour YBF CPUE where environmental variables were offered as covariates selected oxygen concentration, ammonia and turbidity when lagged by different periods (Table 7). The subsequent correlations suggested that decreasing oxygen and increasing ammonia and turbidity may have negatively affected YBF recruitment success. The effect of poor water quality on the mortality of estuarine fish is well established in the overseas literature (see references within Morrison et al. 2009).

Table 7: Environmental variables that had significant correlations (absolute value of the Pearson statistic greater than 0.5) with YBF survey YCS and adult CPUE abundance trends in the Manukau Harbour. Shaded variables were selected by both YCS and CPUE correlations for similar months.

	YCS months having a sig. Pearson correlation		Adult CPUE index months having a sig. Pearson correlation	
Environmental variable	< -0.5	> 0.5	< -0.5	> 0.5
Dissolved Oxygen	-	Aug-Oct	-	Sep
Turbidity	Aug-Sep	Nov-Dec	Jul	-
SOI	Jul-Nov	-	-	-
Dissolved ammonia	-	-	Nov	-

A negative correlation with the SOI implies that stronger YCS are observed in years with more negative SOI values (i.e. El Nino conditions). New Zealand generally experiences stronger patterns of westerly winds during El Nino conditions which may result in enhanced nutrient transfer to west coast harbours leading to higher productivity. Stronger flounder YCS in west coast harbours during El Nino years are consistent with this hypothesis.

Considering the short-life cycle of New Zealand flatfish species, highly variable abundance is expected. Consistent and declining trends in abundance (which can be indicative of recruitment overfishing) should therefore be unlikely (because the life history of flatfish species should allow population recovery when conditions are suitable). The declining trend in FLA CPUE observed in some of the sub-stocks of FLA1 (Kendrick & Bentley 2011) is therefore very likely to be due to factors other than fishing. In light of the correlations with environmental data that are presented here, decreasing water quality may be contributing to reduced flounder abundance. In addition to flounder, other estuarine or estuary-dependent species (e.g. rig, grey mullet and snapper) may also be detrimentally affected by declining water quality. Our overall recommendation is, therefore, that a broader study of trends in water quality, at least for the west coast harbours, be undertaken and potential water quality mitigation measures investigated.

5. ACKNOWLEDGEMENTS

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7. APPENDICES

Appendix 1: Description of Auckland Council Water Quality Variables (text provided by Jarrod Walker Auckland Council).

Water Temperature

Water bodies generally show seasonal patterns in temperature that are correlated with air temperature. Heat transfer between the atmosphere and water surface primarily influences water temperatures of large water masses. Stream temperatures, in the absence of industrial discharges of heated water, are primarily regulated by the extent of riparian vegetation shading of the waterway. In catchments developed for urban uses or intensive agriculture, riparian vegetation has generally been removed to ameliorate flooding problems or maximise land use and as a result stream temperatures tend to be elevated. Shallow coastal saline water temperatures are most commonly influenced by water passage on incoming tides over intertidal sediments that have been warmed by the sun, resulting in an increase in water temperature. Lake water colour can also influence water temperature, because incident radiation is absorbed to a greater extent by darker water bodies resulting in an increase in water temperature.

Elevated water temperature can influence aquatic biota in the following ways:

(i) Community structure in compromised waterways may be dominated by thermotolerant species that can survive fluctuations in temperature, particularly those experienced in summer.

(ii) An increase in water temperature results in a reduction in the dissolved oxygen-carrying capacity of the water. This may be critical for sensitive organisms particularly where saturation levels are already reduced (see next section).

Dissolved Oxygen Saturation

Dissolved oxygen saturation (DO %sat.) gives a direct measure for the assessment of a waterway's ability to support aquatic life and is therefore one of the more important water quality parameters measured in our surveys. However where low saturation levels occur there is often a multiplicity of possible causes.

DO (% sat.) levels show natural fluctuations both diurnally (throughout the day) and seasonally. Diurnal changes are caused predominantly by the respiratory activities of aquatic biota, particularly plants. Seasonal variations mainly follow changes in temperature, which are inversely related to oxygen solubility.

Dissolved oxygen levels around 5 mg/L are known to be stressful to sensitive aquatic biota. This concentration equates to a DO (% sat.) of 40%–60% at the range of temperatures commonly found in Auckland waterways. If low DO (% sat.) levels persist for any extended period of time some organisms that cannot move away may die. Ultimately the diversity of aquatic biota may be reduced to include only those species tolerant of low DO (% sat.).

Amelioration of low DO (% sat.) levels can be achieved by a reduction of point source and non-point source runoff by the modification of land use practices. Riparian vegetation has a role to play in filtering out diffuse sources of oxygen-demanding substances in rural and urban runoff, reducing temperatures and restricting in-stream plant growth by shading. Urbanised areas have the potential to reduce the input of oxygen demanding substances by utilising various stormwater treatment initiatives. In terms of point source inputs, ARC rural and industrial pollution abatement activities are designed to eliminate unauthorised discharges and control authorised discharges of contaminants to a level that can be assimilated by the water body concerned.

In catchments with agricultural development, substantial volumes of stream water are abstracted for irrigation purposes. Consequently DO (% sat.) levels may be further compromised by discharges of pollutants during the summer when the stream assimilation capacity is reduced by such abstractions. Supersaturation of water is not unusual where aquatic plants in the form of macrophytes, periphyton or free-floating algae are abundant. During the hours of daylight the release of oxygen during photosynthesis augments the transfer of oxygen through the surface of the waterbody by diffusion. The negative side to the presence of these plants is the consumption of oxygen at night (i.e. by respiration), which can lead to serious oxygen depletion and subsequent effects on other biota. Depression in DO (% sat.) levels caused by this phenomenon is usually greatest in the early hours of the morning.

Dissolved Oxygen/Temperature Profiles

In lake surveys dissolved oxygen (DO)/temperature profiles are used to determine if there is thermal stratification. A boundary zone exists where temperature decreases by 1 degree Celsius per vertical metre of water depth. Profiles are measured *in situ* using a dissolved oxygen meter and probe lowered from a boat. Stratification occurs during the summer months when calm and warm weather causes the formation of a warm, less dense, water layer on the surface. This layer, the epilimnion, floats on the colder denser water, the hypolimnion, and is separated by the thermocline, where temperature, density and oxygen levels change rapidly with depth (often termed the metalimnion). Low DO levels can have a profound influence on the distribution of organisms in the lake.

Stratification effectively seals off the deeper regions from oxygen replenishment via the lake surface. Hypolimnetic oxygen consumption, resulting from respiration of various microorganisms during the degradation of oxygen-demanding substances in the bottom waters of the lake, can cause DO levels to drop to near zero. A consequence of the low DO levels can be release of nutrients that were formerly bound up in the sediments, into a biologically available form.

In the early autumn when weather patterns become less stable, with cooler temperatures and more wind, sufficient energy in the form of turbulence, convection and advection are provided to effect the mixing of the various stratified layers, a process termed the "turnover". Turnover results in the transport of bioavailable nutrients from the hypolimnion to the epilimnion and, depending on lake water clarity, potentially into the photic zone where they are accessible to phytoplankton. If turnover occurs during the mid-summer period the nutrient liberation can lead to nuisance algal growth.

Biochemical Oxygen Demand

Biochemical oxygen demand (BOD) is a measure of the amount of oxygen required to break down the organic matter in a set volume of water in a five-day period at 20 degrees Celsius. High BOD levels in water bodies indicate the presence of organic matter, which may exert an oxygen demand resulting in a reduced dissolved oxygen concentration and therefore a reduction of water quality. A yardstick for comparison is that waters with a BOD greater than 5 mg/l are considered polluted. Measures available to reduce BOD input have been canvassed in the section on DO (% sat.).

Conductivity

Conductivity is used to estimate the total dissolved solids (soluble salts) content of the water. The soluble salts concentration is an important consideration in relation to abstraction of water for horticultural use and in extreme situations the suitability of water for stock use.

Chloride

The major natural source of chloride is from groundwater, which in the Auckland Region ranges from 17–40 mg/L depending on the geology concerned. High chloride levels are present in wind-blown spray in coastal environments and in rural and urban wastewater. Thus, high chloride levels are often used to indicate the presence of other contaminants in freshwater aquatic systems. Chloride is particularly mobile in groundwater, being weakly adsorbed onto soil particles and readily leached by infiltration rainfall.

High chloride levels in freshwaters have the potential to compromise water use for irrigation of crops or for potable supply. Guidelines for crop watering by foliar application recommend maximum levels of less than 100 mg/L (ARWB TP 41, 1987). The New Zealand Ministry of Health Department guideline value for chloride (recommended upper limit for the avoidance of taste and corrosion problems) in water used for human consumption is 250 mg/L (MoH, 2000).

pH

The pH is a measure of the hydrogen ion concentration and therefore indicates the acid or alkaline nature of the water. The pH range is from 0–14 and each unit represents a ten-fold change in hydrogen ion concentration. Natural freshwaters have a pH of around 7 although 6–9 is considered within the normal range. By comparison seawater is strongly buffered and even small pH changes are significant. The normal saline range is considerably narrower than freshwater; from pH 7.8 to 8.3.

In the absence of contaminant discharges the major influence on pH levels is likely to be the photosynthetic activity of aquatic plants. This occurs when carbon dioxide is absorbed upsetting the carbon dioxide-bicarbonate equilibrium of the stream waters and elevating pH. This problem is most likely to occur in waterways with high nutrient levels and little overhanging vegetation to limit light levels and thereby in-stream plant growth.

pH does not have a directly toxic effect on aquatic biota although many species are not tolerant to wide fluctuations in pH. The principal influence of pH is on the toxicity or mobility of other contaminants present in the water column or sediments. In urbanised situations, receiving water sediments may contain large amounts of heavy metals such as zinc, copper and lead from road stormwater runoff. Decreases in pH would tend to mobilise some of these bound contaminants. The toxicity of other contaminants such as ammonia, which is elevated in some rural waste discharges, generally increases with higher pH and temperature.

Chlorophyll *a*

Chlorophyll *a* level is a measure of the biomass in terms of photosynthetic algae (phytoplankton) abundance. Phytoplankton are microscopic plants which drift freely in the currents of lakes and saline waters. They can determine the suitability of natural waters for a variety of uses. The Lake Managers Handbook (MWD 1987) states that in high concentrations phytoplankton can:

- decrease water clarity;
- alter the colour of the water;
- be toxic to stock and wildlife;
- form unsightly surface scums;
- produce unpleasant tastes and odours;
- alter the water pH;

deplete oxygen through respiration and decay;
water intake filters; and
disrupt flocculation and chlorination processes in water treatment plants.

Chlorophyll *a* level is used in conjunction with total nitrogen and total phosphorus levels, to assess the trophic status of water bodies, particularly lakes. The Lake Managers' Handbook (MWD 1987) defines lake trophic status in terms of average annual chlorophyll *a* level and annual maximum chlorophyll a level as follows:

Average annual (μg/L)	Maximum annual (μg/L)	Trophic status
<2	<5	Ultra-oligotrophic (ultra low enrichment)
2–5	5–15	Oligotrophic (very low enrichment)
5–15	15–40	Mesotrophic (medium enrichment)
>15	>40	Eutrophic (highly enriched)

Water Clarity

Public perception of water quality is often based on their observation of water quality or clarity, in that poor water clarity is aesthetically unpleasing, regardless of other water quality parameters. In the ARC baseline water quality monitoring programmes water clarity is expressed by measurements of turbidity, black disk transparency and Secchi disk depth. The critical measures of acceptable water clarity are: for recreational waters clarity greater than 1.6 metres as measured by the black disk technique, and for aesthetic purposes no significant change. A significant change is considered to be a 20% change in black disk reading.

Turbidity

Turbidity is a measure of the passage of the degree to which light is scattered in water by suspended particles and colloidal materials. Samples are analysed in the laboratory using a meter and the results are given as nephelometric turbidity units (NTU). When turbidity levels are high, light penetration is reduced, thereby limiting the ability of aquatic plants (algae and macrophytes) to photosynthesise (i.e. a reduction in the so-called euphotic depth). Organisms that are visually oriented may have difficulty locating and catching prey in turbid water and the fine suspended material that is characteristic of turbid water may detrimentally affect gill structures of aquatic organisms.

Black Disk Transparency

Black disk transparency is a measure of horizontal water clarity. The black disk reflects very little light and black disk transparency is the distance at which it becomes visible to an observer (using an underwater viewer). It is a good estimate of the distance that sighted animals can see horizontally under water.

Secchi Disk Depth

Secchi depth is a measurement of vertical optical water clarity, which is a function of light penetration. It is usually applied in deeper water bodies such as lakes. The light penetration defines the depth to which photosynthetic plants can survive, known as the euphotic depth. As a rough guide, the euphotic depth is taken as 2.5 times the Secchi depth which is the depth at which a quartered 200 mm diameter black and white disk becomes visible to an observer as it is raised through the water column.

Suspended Solids (also called non-filterable residue)

Suspended solids (SS) is a measurement of the suspended material in the water column, including plankton, non-living organic material, silica, clay and silt. High SS levels reduce light penetration and provide media for pollutants to attach to, resulting in a reduction in water quality for a variety of uses, such as horticulture, irrigation, stock water supply, and recreational and ecological functions. Under the appropriate conditions the suspended material can settle out as sediment thereby causing further problems, such as smothering of biota.

SS burdens to waterways can be reduced in a variety of ways depending on the type of land use concerned:

- In rural catchments riparian zone management provide an effective filter for diffuse sources of SS and reduces streambed and bank scouring by dissipating the energy of floodwaters.

Preventing stock access to stream beds and banks is a useful mitigation tool for reducing excessive SS.

- In urban and industrial areas SS can be reduced through the implementation of storm water control measures. The period when land is being urbanised has the greatest potential to mobilise sediments to waterways. ARC Environment has produced urban earthworks guidelines to minimise SS runoff from exposed erodible soils.

Microbial Indicators

Microbial indicator organisms are typically used in water quality monitoring to provide a measure of faecal contamination and hence the sanitary quality of water resources. A number of different indicator organisms and monitoring strategies can be used depending on whether the purpose of sampling is simply to detect and quantify the level of contamination, or whether some measure or index of public health risk is required.

The indicator organisms used for water quality monitoring are generally bacteria that are present as normal inhabitants in the gut of healthy warm-blooded animals, including humans, and are shed in large numbers in faecal matter (at a rate of 10^6 – 10^9 per gram). They are not usually considered to present a risk to public health when present in natural waters (i.e. they are not generally disease causing or pathogenic when contacted through this route), but their presence is taken to indicate faecal contamination and hence the possibility that pathogenic microorganisms that are found in the gut may also be present.

It is necessary to use indicator organisms for routine monitoring purposes because there is such a wide variety of pathogens that may be present in faecal matter, that it is impossible to test for all of them at once. Detection of some pathogens, particularly viruses, is also expensive and time consuming. Also, the infective doses for many pathogens, particularly of viruses, are so low as to make routine measurement impracticable.

In New Zealand three bacterial indicator groups have been routinely used for water quality monitoring. These are the presumptive coliform, faecal coliform, and enterococci groups.

Coliforms or Presumptive Coliforms

The term coliform is used to describe a heterogeneous group of bacteria belonging to the family *Enterobacteriaceae*, which are characterised by their ability to ferment lactose with the production of acid and gas at 35°C. Included within this definition are members of the *Escherichia*, *Klebsiella*, *Enterobacter*, *Serratia*, and *Citrobacter* genera. While members of all of these genera are typically found in faecal material, only one, *Escherichia coli*, is truly faecal specific.

The results of coliform or presumptive coliform tests are often highly variable and do not necessarily indicate the degree of faecal contamination present in a waterway. This is because members of the coliform group are also found as natural inhabitants of soil and decaying vegetation, and therefore elevated levels in waters may be due to naturally occurring organisms. Nevertheless, the presumptive coliform test may provide useful information on the level and nature of contamination when used in association with other analyses such as the faecal coliform test.

Faecal Coliforms

Faecal coliforms represent a subset of the coliform group that are differentiated by their ability to ferment lactose with the production of acid and gas at the elevated temperature of 44.5°C. This group are more specific indicators of faecal contamination than the coliform group, although the functional definition still includes some organisms that are natural inhabitants of soil and decaying vegetation.

The use of the term faecal in the group description is therefore somewhat misleading, and has lead to the use of the term "thermotolerant coliforms" as an alternative group name.

Faecal coliforms have historically been the indicator of choice for assessment of the sanitary quality of natural waters and have formed the basis of the previous microbiological guidelines for recreation and shellfish growing waters. However, studies undertaken on behalf of the United States Environmental Protection Agency (USEPA) comparing indicator levels with health effects have indicated that enterococci (see later) provide a much better index of health risk than faecal coliforms. The USEPA have subsequently developed enterococci guidelines for health risk monitoring of recreational water quality in the USA. These guidelines have been used to form the basis of New Zealand's provisional guidelines for recreational water quality monitoring. For further information on this topic refer to the "Recreational Water Quality Guidelines" published by Ministry for the Environment and Ministry of Health, Wellington, November 1999.

However despite this the faecal coliform group is still considered appropriate for qualitative monitoring of faecal contamination in natural waters, and for assessment of long terms trends in water quality over time. It is in this context that the indicators are used in the baseline water quality studies. The only major impediment to this use is the inability to discriminate between contamination of human and non-human origin. Such assessments must be made on the basis of subjective evaluation of likely sources and routes of contamination within the catchment.

Enterococci

Members of the genus *Enterococcus* comprise another group of bacteria that are found in the gut of warm blooded animals and are commonly used as a health related indicators of saline recreational water quality. Enterococci analysis is typically carried out by the membrane filtration method using mE and EIA media. This method is selective for two species, *Ent. faecalils* and *Ent. faecium,* which are prominent in human faecal matter, although other faecal and non-faecal associated enterococci may also be detected using this method. Interpretation of results and assessment of public health risk therefore requires that consideration be given to the likely sources of contaminants.

Nutrients

Nutrients are chemical compounds that are necessary for normal plant growth and are divided loosely into macro- and micro-nutrients. Routine water quality monitoring records two groups of essential macro-nutrients.

The availability of readily assimilated forms of the nutrients nitrogen and phosphorus are commonly accepted as factors limiting aquatic plant growth. Anthropogenic activities increase the nutrient loading through the discharge of waste products, fertilisers and general storm-water runoff. Nutrient enrichment can result in a proliferation of algae and macrophytes in waterways, which potentially has a number of detrimental effects including:

- Choking waterways leading to reduced drainage capacity,
- Loss of amenity values,
- Physical habitat reduction,
- Excessive fluctuations in dissolved oxygen and pH,
- Reduced suitability for stock watering or horticultural irrigation.

The adverse effects of elevated nitrate levels can be mitigated by the provision of riparian vegetation providing sufficient shading to preclude or minimise in-stream plant growth. Riparian vegetation also provides a mechanism for intercepting contaminants by filtering direct runoff and uptake of nitrate from the soil at the ground water interface. The proactive approach is to prevent or minimise the discharge of nutrient rich discharges into waterways. Nutrient levels entering waterways can be reduced by a number of land management options including;

- Limiting concentrations from point sources by consent conditions,
- Requiring land application of wastes in a way that minimises subsequent input to streams,
- Implementing land management techniques such as riparian zone protection to reduce diffuse input.

Ammonia

Ammoniacal nitrogen is a macro-nutrient but is considered in general water quality evaluations in terms of its toxicity to many aquatic animals. Ammonia occurs in a number of waste products, which if discharged to the environment can result in elevated ammonia levels. Ammonia is reported as a combination of un-ionised ammonia (NH_3) and the ammonium ion (NH_4^+). At normal pH values the latter form predominates. Un-ionised ammonia is the more toxic form to aquatic life. The toxicity of ammonia is very dependent on water temperature, salinity and pH (USEPA, 1999). Regulatory agencies, such as the ARC Environment, have tended to rely on overseas criteria such as those promulgated by the USEPA. The ARC has commissioned studies on Auckland freshwater biota, which corroborate that USEPA criteria are appropriate.

Ammonia toxicity for given pH and temperature combination can be calculated using a mathematical equation. As a generalisation a chronic or long term exposure limit of 0.77 mg/L is appropriate for sensitive freshwater organisms under ambient conditions. In saline waters ammonia toxicity is influenced by salinity in addition to pH and temperature. The chronic exposure limit for sensitive saline organisms under ambient conditions is 2.3 mg/L.

Long term or chronic effects on biota include the limitation of species that can survive in the waterway to those tolerant of ammonia. In addition sublethal effects, such as disruption of feeding patterns and removal of food sources, reduction of reproductive viability and restricted recruitment of juvenile organisms in response to long term exposure to ammonia, have been documented by the USEPA.
In catchments with intensive farming practices ammonia rich wastewaters can come from several sources. Potential causes of diffuse input include rainfall on areas adjacent to waterways that have been grazed, had spray irrigated with wastewater or had fertilisers such as ammonia urea applied to them recently. Rural point sources include race runoff, oxidation pond discharges, silage leachate, or raw wastes when disposal systems break down or are not used as intended.

Nitrite

Nitrite is the intermediate step in the conversion of ammonia to nitrate. It is usually short lived in the aquatic environment in the presence of oxygen and is therefore indicative of a source of nitrogenous waste in the immediate vicinity of the sampling site. It is intermediate in toxicity to ammonia and nitrate (USEPA, 1985).

Nitrate

Nitrate is the end product of the breakdown (oxidation) of ammonia through the intermediate step of nitrite by microbial decomposition. It is not particularly toxic to aquatic life (USEPA, 1985). Water for use as potable supply is limited to 10 mg/L on public health grounds. In terms of crop irrigation water requirements higher nitrate levels could be seen as an advantage saving on fertiliser costs. For stock drinking water requirements the recommended limit is 100 mg/L.

Sources of nitrate in aquatic systems are similar to those discussed for ammonia. Nitrate is poorly bound to the soil and is therefore highly mobile. It is readily leached into local groundwater systems, particularly under high rainfall events. In winter when ground conditions become saturated the capacity of the soil to assimilate waste is reduced, resulting in elevated nitrate levels in runoff.

Nitrate is an important plant nutrient (which is generally non-limiting), which in conjunction with sufficient available phosphorus can lead to proliferation of aquatic plants (algae and macrophytes).

Respiration of aquatic plants at night can lead to reductions in dissolved oxygen to the point that other aquatic organisms may become stressed or killed. Photosynthetic activity of aquatic plants also leads to elevated stream pH, which has an effect on the toxicity of other contaminants in the water such as ammonia.

Total Kjeldahl Nitrogen

Total Kjeldahl nitrogen (TKN) is a measure of the organic nitrogen plus ammonia concentration of a water sample. It includes such natural materials as proteins and peptides, nucleic acids and urea and numerous synthetic organic materials. It is used in this report to calculate the total nitrogen content of water samples.

Total Nitrogen

Total nitrogen is the combination of nitrate, nitrite and TKN, it is used to estimate the "bioavailable" fraction of nitrogen in waterways. It is also used in conjunction with total phosphorus and chlorophyll a levels, to assess the trophic status of water bodies, particularly lakes. The Lake Managers Handbook (MWD 1987) defines lake trophic status in terms of average annual total nitrogen level as follows:

Total Nitrogen (mg/L)	Trophic status
<0.2	Ultra-oligotrophic (ultra-low enrichment)
0.2– 0.3	Oligotrophic (very low enrichment)
0.3– 0.5	Mesotrophic (medium enrichment)
>0.5	Eutrophic (highly enriched)

Total Phosphorus

Total phosphorus is a measure of all the phosphorus present in the sample and includes the soluble (bioavailable) fraction that is adsorbed onto sediment particles and present in the form of algae and other organic matter. The Lake Managers Handbook (MWD 1987) defines lake trophic status in terms of average annual total phosphorus level as follows:

Total Phosphorus (mg/L)	Trophic status
<0.01	Ultra-oligotrophic (ultra-low enrichment)
0.01–0.02	Oligotrophic (very low enrichment)
0.02–0.05	Mesotrophic (medium enrichment_
>0.05	Eutrophic (highly enriched)

Dissolved Reactive Phosphorus (soluble reactive phosphorus)

Dissolved reactive phosphorus (DRP) is considered to be the bioavailable fraction of phosphorus and is an important as an indicator of water quality. It is frequently cited as the nutrient limiting the proliferation of algae and other aquatic plants in New Zealand waterways. Levels required to stimulate in stream plant growth are reportedly as low as 0.01 mg/L.

Appendix 2: Combined length frequency of YBF sampled from Manukau Harbour commercial setnet catches in 1987 (Ministry for Primary Industries unpublished data)