Chapter 2: Work Package 1A: Establish relationship between the EPA Quality Rating System (Q-value) and Fish stock composition and abundance

2.1 Overview of chapter

This chapter deals with the relationship between fish and ecological quality as indicated by the Quality Rating System (Q-value). An overview of the main methods used throughout the project is given. Variation in fish populations over a range of Q-values was assessed for a large number of river sites. This work package also investigated whether classification using macroinvertebrate data could be applied to fish abundance data and whether both taxonomic groups responded similarly to certain environmental variables.

2.2 Introduction

Biological communities or assemblages of similar organisms have been generally recognised as useful in assessing water quality because they are sensitive to low-level disturbances and function as continuous monitors. Both fish and benthic macroinvertebrates have been used in water quality assessment. There has been little agreement on which group is the more effective (Berkman and Rabeni, 1986), however in recent years macroinvertebrates are widely accepted as the most useful – the fact that macroinvertebrates are the primary biological element currently being used in the WFD Intercalibration process supports this contention (Martin McGarrigle, *pers. comm.*).

The EPA Quality Rating (Q-value) System has been used to monitor the ecological quality of streams and rivers in Ireland since 1971. There is a tendency for higher quality sites (Q5 and Q4-5) to be located in soft-water areas as indicated by alkalinity values. It is almost axiomatic that our cleanest rivers are in areas of low population density and low agricultural intensity (McGarrigle, 2001). An alarming decrease in high quality sites has been reported (Champ, 2000) and this is associated with an increasingly enriched condition of Irish rivers with a five-fold increase in slight pollution (Q3-4) and a three-fold increase in moderate pollution between 1971 and 1997 (Bowman and Clabby, 1998). Slightly polluted sites (Q3-4) are highly eutrophic, water quality is unsatisfactory and algal cover is often extensive (McGarrigle *et al.*, 2002). These slightly polluted locations may still have plenty of fish and salmonids may occur but the range of diurnal oxygen variation can be greatest at these sites which are most at risk of fish kills (McGarrigle, 2001)

The species composition, number, and age structure of fish populations which occur in any river varies from location to location (spatial), seasonally and annually (temporal). A variety of physical, chemical and biological factors influence this variability (Huet, 1959; Hynes, 1970; Karr, 1981 and 1991; Fausch *et al*., 1984 and 1990). In an attempt to examine natural spatial and temporal variation in fish communities, four core rivers were selected, (Robe, Rye Water, Dunkellin and Oona Water), wherein, various sites were sampled for fish stock assessment on two or more occasions in 2001, 2002 and 2003.

These rivers exhibit varying degrees of biological and physical impairment due to a variety of anthropogenic pressures. Evaluating fish stock variations based on few repeated sampling occasions at selected sites is of limited value and could be erroneous (Elliott, 1994) when little is known about the spatial and temporal variations in the 'baseline' or equilibrium density. To overcome this, additional fisheries datasets for river channels at high ecological status (Q5 and Q4-5), on the River Slaney catchment, were also assessed to obtain essential information on natural temporal variations in fish stock density (around the natural 'mean density') for each location in the absence of environmental impact, over a period of ten years.

Eutrophication is acknowledged as the main cause of water quality degradation in Ireland (McGarrigle, 1998) and the effect of this on the fish species composition and abundance in rivers in Ireland was assessed by surveying sites with a wide range of Q-value scores. There are many methods available to assess eutrophication by examining community structure based on single taxonomic groups such as macroinvertebrates, fish, algal communities, diatoms and macrophytes (Kelly and Whitton, 1998). These groups are often used to reflect change across the whole community. Few studies have been carried out to determine concordance between these taxonomic groups in lotic environments (Kilgour and Barton, 1999; Paavola *et al*., 2003). Community concordance is the degree of similarity in the pattern of community structure between taxonomic groups sampled in the same location (Paskowski and Tonn, 2000). Paavola *et al*., (2003) found no concordance between fish, benthic macroinvertebrates and bryophytes in 32 headwater streams in Finland because each taxonomic group responded to different environmental variables, the macroinvertebrates related to stream size and pH and the fish were related to depth, substrate size and water oxygen content. McDonnell (2005) found that fish and macroinvertebrates, from a subset of sites from this project (51 sites), responded similarly to variables such as velocity and percentage instream vegetation and concordance was observed between the two biotic groups.

2.2.1 Aims of workpackage

The primary aim of this work package was to assess the impact of water quality, as evidenced by the EPA's Quality Rating System, on riverine fish stocks to establish if a relationship exists between fish and quality ratings by investigating fish species composition at sites of varying Q-values (Q1 to Q5), and to assess the feasibility of using fish assemblages as biological indicators of river water quality in Irish rivers. The physical environment obviously plays an important role in controlling the distribution of fish species and community composition along a river from high to low gradient (Huet, 1959; Welcomme, 1985). Thus the effects of water quality are effectively layered on top of the underlying physical and hydromorphological factors in the statistical models. In doing this it was also necessary to separate the effect of physical habitat from water quality impacts.

2.3 Methods

2.3.1 Site selection

This work package involved one-off electric fishing surveys at 374 river sites between 2001 and 2003. Initially (2001) the project drew heavily on fish stock assessment surveys being conducted by the Central and Regional Fisheries Boards as part of fisheries catchment management programmes on the River Laune (33 sites), Lough Melvin (14 sites) and similar projects conducted by the Boards in Donegal (Rosses, 16 sites), Kerry (Feale, 22 sites), Meath (Boyne, two sites), Mayo (Moy, four sites) and Cavan (Sheelin, five sites). This work was additional to, and independent of, the survey work conducted on other core rivers as part of the project. In 2002 and 2003 additional sites were selected to cover as wide a range of environmental variables and EPA Q-values as possible, particularly Q1, Q2, Q2-3 and Q5. Published EPA Q-values were used to select potential sites which represented a nutrient and water quality gradient. A further 145 sites (where the necessary parameters, i.e. fish, macroinvertebates and abiotic variables, were contained in the datasets) were extracted from the CFB archival database for use in the project. Figure 2.1 indicates the location of all sites included in the project database (519 sites, 15 of which were sampled on 3 occasions and 47 sites on two occasions). A specific effort was made in 2003 to survey potential reference sites (Q5) selected for the Water Framework Directive.

2.3.2 Core rivers

Four rivers (Oona Water-13 sites, Ryewater-2 sites, Dunkellin-15 sites and the Robe-16 sites) where scientific investigations were ongoing or had previously been undertaken, were selected for study. Various sites were sampled for fish stock assessment on two or more occasions in an attempt to examine temporal variations in fish populations.

The Oona Water

The Oona Water is a tributary of the Blackwater River, one of the principal influent rivers of Lough Neagh, in Northern Ireland drains a catchment area of 107 km². The Oona Water was arterially drained in the 1980s. The quality of the main river is monitored at three locations by the Environment and Heritage service (EHS) (O'Neill and Hale, 1998 and 2000). Diffuse transfer of agricultural phosphorus to freshwater is particularly pervasive in the catchment and this was the subject of a separate EPA ERTDI project (2000-LS2.1.1a-M1) (Jordan *et al.*, 2005).

The Rye Water

The Rye Water is the main tributary of the river Liffey, rises in Co. Kildare and drains a catchment of approximately 100km^2. It receives nutrients from various point and diffuse sources which maintain the river in a eutrophic state. The river was altered hydrologically, (Kelly and Bracken, 1998) and fisheries rehabilitation (de-silting and instream diversity restored) was carried out downstream of Sandford's Bridge in 1993 (Kelly, 1996). Fish and macroinvertebrate assemblages have been monitored in this section of the river since 1992.

The Dunkellin River

The Dunkellin river (also referred to as the Kilcolgan River) drains about 400 km² in south Co. Galway and discharges to Galway Bay. Previous studies had been undertaken on the ecology of the river's aquatic invertebrates and fish assemblages (Buckley, 1993; Callaghan, 1993). Water chemistry, macroinvertebrates and fish communities were surveyed at 16 and 15 sites site in 2001 and 2002 respectively.

The Robe River

The Robe rises west of Ballyhaunis and flows into Lough Mask downstream of Ballinrobe and is the largest of the Lough Mask spawning tributaries (63.6km in length). The Robe drains a catchment area of approximately 320 km^2. The geology of the catchment is predominantly carboniferous limestone. The river Robe was the subject of a major arterial drainage scheme during the 1980s. Seventeen, twenty and fourteen sites were electric fished in the Robe catchment in July/August 2001, 2002 and 2003, respectively.

The Slaney River

The Slaney River is 117.5 km long and drains a catchment area of 1762km². The river is eutrophic in its lower reaches. The Central and Eastern Regional Fisheries Boards have conducted fish stock surveys at selected sites in the Slaney catchment intermittently since 1991. The same sites were studied in August/September at two or three year intervals. Concurrent analysis of macro-invertebrates was not conducted during these investigations. However, the Slaney River and its tributaries are surveyed by the EPA as part of the long term national biological monitoring programme (Clabby *et al.*, 2002). Five fish survey sites were selected adjacent to EPA water quality monitoring sites.

2.3.3 Sampling regime

A standard methodology was drawn up for the project based on procedures developed by the CFB and EPA over a number of years and incorporated common practices normally employed by the various partners (Kelly, 2001). The standard methodology included fish, macroinvertebrates and hydrochemistry sampling, and a physical habitat survey. A standard survey sheet was provided to all partners (Appendix 3). Surveys were carried out between July and September (to facilitate the capture of 0+ salmonids) when stream and river flows were moderate to low. Standard EPA methods were used to assess Q-values (Clabby *et al.*, 2001). This latter task was made possible through the organisation of a workshop in June 2001 at which instruction was provided by EPA personnel.

Fish Stock Assessment

All fish sampling methods are generally considered selective to some degree, however, electric fishing has proven to be the most comprehensive and effective single method for collecting stream fish (Barbour *et al.*, 1999). Therefore, electric fishing was the method of choice in this project to obtain a representative sample of the fish assemblage at each sampling site. The technique complied with CEN guidance for fish stock assessment in wadable rivers (CEN, 2003).

Each site was sampled by depletion electric fishing involving one or more anodes, depending on the width of the site. Each sampling area was isolated using stopnets or was clearly delineated by instream hydraulic or physical breakpoints such as well defined shallow riffles or weirs. A number of fishings (i.e. two to four) were carried out in the contained area. In small shallow channels (<0.5-0.7m in depth), a portable landing net (anode) connected to a control box and portable generator (bank-based) or electric fishing backpack was used to sample in an upstream direction. In larger deeper channels (>0.5-1.5m), fishing was carried out from a flat-bottomed boat(s) in a downstream direction using a generator, control box and a pair of electrodes. All habitats, in wadable and deeper sections, were sampled (i.e. riffle, glide, pool).

Fish from each pass/fishing were sorted and processed separately. All fish species present were measured for length and weight and scales for age analysis were removed from a subset of samples. Fish were measured for fork length in 1cm length groupings. All fish were held in a large bin of water after processing until they were fully recovered and then returned to the river.

Population estimates were calculated using the two fishing depletion of Seber and Le Cren (1967) or the three fishing depletion method of Zippin (1958) or Carle and Strub (1978). Minimum densities were calculated where single fishing was carried out (Crisp, *et al.*, 1974). All population estimates were converted to minimum densities (i.e. number m^{-2} for combined runs) to standardize the dataset for statistical analysis.

Macroinvertebrates

Macroinvertebrates were collected at each site, (after setting stop nets) before electric fishing, using a two and a half minute kick sample from a riffle. By working across the river, all available habitats were sampled and in addition to kick sampling, stone washing and weed sweeps were carried out. The macroinvertebrates were transferred to a white sorting tray, examined and Q-values determined. To ensure an element of quality control some samples were preserved in 70% methanol and retained for examination. In the laboratory samples were rinsed using a 500µm mesh sieve to remove fine sediment and placed in a white tray.

Macroinvertebrates were sorted and identified to family/genus/species depending on the EPA requirements for Q-value determination. The relative abundance of each taxon identified was recorded and an EPA Q-value rating was applied (see Appendix 1) to each site sampled as an indication of water quality.

Environmental/abiotic variables

An evaluation of habitat quality is critical to any assessment of ecological integrity and was performed at each site at the time of biological sampling. Physical characterisation of a stream includes documentation of general land use, description of the stream origin and type, summary of riparian vegetation and measurements of instream parameters such as width, depth, flow and substrate (Barbour *et al*., 1999). A number of habitat variables were measured at each site to complement the species lists (Table 2.2).

At each site the percentage of overhead shade, percentage substrate type and instream cover were visually assessed. Land use, altitude, catchment area, eastings and northings, distance from source and sea and stream order (Strahler, 1952) were calculated using Arc View 3.2 and Digital Terrain Modelling (DTM). Water width was measured at three transects along the reach fished. Water depth was measured at five intervals along each transect. The percentage of riffle, glide and pool was measured over each reach surveyed. Riffles were classified as areas of fast water with a broken-surface appearance; pools were areas of slow deep water with a smooth surface appearance and glides were intermediate in character.

Table 2.2 Selection of environmental/abiotic variables measured or calculated at all sites during the project

Environmental variable	Min	Max	Mean	Footnote
Geographic variables				
Altitude (m)	2	263	70	1,2
Catchment area (km^2)	0.0168	1685	32	2
Stream order (Strahler)	1	6	2.76	2
Slope (%)	0	37.49	1.6	2
Dist. from source (m)	86	81361	7019	3
Dist from tidal limit (m)	386	166883	39067	4
Geology	1	19	13	5
River reach sampled				
Length fished (m)	13	410	40.05	6
Mean depth (m)	0.05	1.6	0.23	7
Max depth (m)	0.1	2.6	0.47	8
Mean wetted width (m)	0.8	30	4.2	9
Surface area (m^2)	14.94	7200	206.2	10
Shade due to tree cover (%)	0	100	33.45	11
Bank height (m)	0	6	1.3	11
Instream cover (%)	0	100	22.1	11
Landuse	1	6	-	12,13
Bank slippage	0	1	-	14
Bank erosion	0	1	-	14
Fencing (RHS & LHS)	0	1	-	14
Trampling (R & L)	0	1	-	14
Velocity status	1	3	-	15
Velocity rating	1	5	-	16
Flow type (%)				
Riffle	0	100	36.5	17
Glide	0	100	37.4	17
Pool	0	100	26.2	17
Substrate type (%)				
Bedrock	0	100	4.4	11
Boulder	0	80	11.1	11
Cobble	0	100	35.2	11
Gravel	0	95	29.4	11
Sand	0	80	11.1	11
Mud/silt	0	100	9.3	11

Water chemistry				
Conductivity (uS cm^{-1})	33	847	328.9	18
Alkalinity (meq l^{-1})	0.464	8.588	3.052	19, 20
Total Hardness (mg l^{-1} CaCO3)	3.456	455.6	162.7	20,21

***Footnotes*:**

1. Geographical information system using 1:50,000 scale. Ordnance survey of Ireland vector data
2. GIS – calculated from EPA Digital Terrain Model (DTM)
3. Distance from source – EPA WFDGIS dataset - length of channel upstream of each sampling point to source
4. Distance from tidal limit – EPA WFDGIS dataset – distance from river reach downstream to the coast (tidal limit is defined as the High Water Mark)
5. GIS using Geological Survey of Ireland map
6. Measured over length of site fished
7. Mean of 15 depths taken at 3 transects through the site
8. Measured at deepest point in stretch fished
9. Mean of 3 widths taken at 3 transects
10. Calculated from length and width data
11. Estimated visually at time of sampling
12. Landuse in the immediate area of the site estimated visually at time of sampling (1-pasture, 2-woodland, 3-tillage, 4-urban, 5-bog and 6-other)
13. GIS using CORINE data
14. Bank slippage, bank erosion, fencing estimated visually at time of sampling (presence or absence recorded as 1 or 0)
15. Velocity status=Water levels-estimated visually at time of sampling-3 grades (1-flood, 2-normal & 3-low)
16. Velocity rating-estimated visually at time of sampling-5 ratings given (1-torrential, 2-fast, 3-moderate, 4-slow, 5-very slow)
17. Measured at time of sampling (when measuring length of site)
18. Measured at time of sampling with a YSI multimeter or WTW meter (for conductivity, pH and DO)
19. Alkalinity – Titration with 0.1N HCL, end point pH 4.1
20. Where alkalinity values were missing they were calculated using regression analysis (Appendix 4)
21. Total hardness – Potentiometric titration, Central Fisheries water chemistry laboratory and by calculation using Ca and Mg data measured on an ICP-MS, CFB laboratory

Water chemistry

A water sample was collected from each site and returned to the laboratory for analysis. Thirty-six chemical variables were measured, including total phosphorus, total nitrogen, alkalinity, total hardness, total oxidised nitrogen, molybdate reactive phosphate (MRP) and 20 heavy metals (Table 2.3). Conductivity, temperature, dissolved oxygen and pH, were measured on site at most locations using a YSI multiprobe. Conductivity, total hardness and alkalinity were considered to be important variables in relation to fish abundance and composition. Therefore where gaps for these variables occurred in archival datasets these were filled using regression analysis (Appendix 3).

Table 2.3: Water chemistry variables recorded during the project

	Mean	Minimum	Maximum	No. samples
In situ				
Dissolved oxygen (% sat)	99.9	56.3	168.0	232
Dissolved oxygen (mg l^{-1} O_2)	10.8	4.0	17.0	219
pH	7.7	5.4	8.9	226

Temperature (^oC)	11.7	1.50	21.0	220
Laboratory				
Molybdate reactive phosphate- MRP (mg l^{-1})	0.039	0.001	1.010	225
Total oxidised nitrogen- TON (mg l^{-1})	1.420	0.034	8.541	191
Total Phosphate (mg l^{-1})	0.037	0.008	1.102	161
Total Nitrogen (mg l^{-1} N)	0.986	0.011	3.072	91
Ammonia (mg l^{-1} N)	0.077	0.000	0.372	95
Colour (hazen units)	33.318	0.000	146.600	134
Turbidity	4.065	0.250	50.000	80
Suspended Solids (mg l^{-1})	4.263	0.300	28.400	80
ICP-MS analysis				
Ca (mg l^{-1})	57.750	0.610	217.160	251
K (mg l^{-1})	1.890	0.160	11.720	251
Mg (mg l^{-1})	4.301	0.400	14.760	251
Na (mg l^{-1})	8.716	1.100	31.280	159
Al (µg l^{-1})	20.084	0.000	375.930	272
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Ba (µg l^{-1})	29.413	0.000	125.290	272
Cd (µg l^{-1})	0.743	0.000	30.870	263
Co (µg l^{-1})	5.418	0.000	99.000	272
Cr (µg l^{-1})	1.026	0.000	10.180	271
Cu (µg l^{-1})	1.153	0.000	12.010	266
Fe (µg l^{-1})	109.954	0.000	815.690	117
Mn (µg l^{-1})	9.086	0.000	185.570	266
Mo (µg l^{-1})	93.347	0.000	1717.620	195
Ni (µg l^{-1})	0.976	0.000	8.970	272
Pb (µg l^{-1})	2.834	0.000	47.330	260
Rb (µg l^{-1})	21.335	0.000	719.270	205
Sr (µg l^{-1})	212.940	0.000	851.960	266
U (µg l^{-1})	20.871	0.000	1124.920	195
V (µg l^{-1})	1.185	0.000	166.000	202
Zn (µg l^{-1})	3.008	0.000	125.340	262

Database and GIS development

Geographical Information Systems (GIS) software facilitates the input, management, manipulation, analysis and output of spatial and tabular data. It allows the collection, storage, retrieval, analysis modelling and map presentation of many sources of information including terrain detail, hydrology, water quality, land use, habitats, geology and other sources of geographically distributed phenomena. The results derived can subsequently be represented as geographical referenced outputs such as spreadsheets or maps (McGinnity *et al.*, 1999). It is a powerful spatial analysis tool and is being used increasingly in environmental management and natural resource planning.

The Central and Regional Fisheries Boards utilise a common GIS system which contains a suite of datasets. These datasets include OS raster mapping at scales of 1:10560 and 1:50000, rivers, streams, lakes, catchment boundaries, forestry, geology, CORINE landcover etc. Additional database sources (e.g. EPA water quality data, Department of Agriculture land use information etc.) have been integrated into the Fisheries Board GIS, thus enabling fisheries data to be interpreted in terms of other environmental factors (e.g. the distribution of fish in relation to pollution status, land use etc.).

GIS will be the delivery platform for the WFD, and it has been used extensively during this project. The principal software used was Arcview 3.2 and Spatial Analyst extension (version 2.0). A comprehensive series of datasets, including site photographs, organised into an integrated GIS project utilising Arc View software is a final deliverable from the project.

Quality assurance/Q-value validation

The Environmental Protection Agency (EPA) has produced a detailed specification for macroinvertebrate sample collection and analysis, this procedure was followed by all team members. To complement this a Q-value workshop was organised for all project team members at which EPA biologists provided additional training. When all Q-values were estimated the data were screened and validated by the project manager. An external audit was then carried out by an EPA biologist in collaboration with the project manager before data analysis could begin.

2.3.4 Data analysis

One-way ANOVA was used to test for significant differences between composition and abundance of all fish species (i.e. minimum density, no. m^{-2}) between each Q-value designated group. Percentage data was arcsine transformed. All abundance data were log transformed (log x+1). An LSD post hoc test was used to test which groups were significantly different from each other.

Sites for which macroinvertebrate data were not available were removed from the analysis (406 sites remaining). Rare species (those species occurring at <5% of sites) were also removed from the dataset prior to analysis. Two-way Indicator Species Analysis (TWINSPAN – Hill, 1994), a hierarchical classification programme, was used to classify sites according to their species composition. Any dataset can be classified but it is important to determine the validity of the classification. Validation is usually carried out by assessing whether the within-group differences are less than the between group differences (Dodkins, 2003). Validation of the fish and macroinvertebrate end groups in TWINSPAN was verified using the Multi Response Permutation Procedure (MRPP) in PC-ORD (McCune & Mefford,

1997) using Euclidean distance measures. MRPP is a non-parametric procedure for testing the hypothesis that there are no significant differences between two or more groups and has the advantage over ANOVA that it is not reliant on multivariate normality and/or homogeneity of variances.

De-trended correspondence analysis (DCA – ter Braak and Prentice, 1988) was used as an initial indirect ordination technique. Ter Braak and Smilauer (1988) recommended that a linear model (e.g. PCA) be used with a gradient length of less than 2 standard deviation units, for values greater than this, unimodal methods (such as Correspondence Analysis and Canonical Correspondence Analysis) are best. Latent gradient lengths for both first and second ordination axes exceeded two standard deviations, therefore Canonical Correspondence Analysis (CCA) was used to evaluate relationships between fish and macroinvertebrate assemblages and environmental variables and to organise the taxa along environmental gradients. The significance of the all canonical axes was tested using Monte Carlo permutation tests (with 9999 permutations). The taxa scores and the correlations between environmental variables and the axes were plotted as biplots. The lengths and positions of the arrows provide information about the relationship between the environmental variables and the derived axes. Arrows that are parallel to an axis indicate a correlation and the length of the arrow reflects the strength of that correlation. The position of the sites/species indicates their status with respect to the environmental variables (Ter Braak, 1986).

The TWINSPAN end groupings for one taxonomic group were imposed on the other dataset and vice versa to determine whether the classification for one group would apply to the other. Concordance between macroinvertebrates and fish was assessed using Analysis of Similarity (ANOSIM) (Primer V.5). ANOSIM tests the Null hypotheis that there are no assemblage differences between groups of samples (such as TWINSPAN or Q-value groups). As a final test of concordance both groups were compared using a Mantel test (Clarke and Gorley, 2001).

2.4 Results

A total of 519 sites were included in the database, however, only 470 of these have been subjected to univariate analysis and 394 to multivariate analysis due to incomplete datasets. The number of sites electric fished by each partner (including archive sites) ranged from 17 to 312 (Table 2.4). Fish were electric fished at sites in first order to sixth order streams at a range of Q-values (Table 2.5). All, but one site (UCD068-Vartry tributary at 263m) were located below 250m altitude.

Table 2.4: Number of sites electric fished per organisation (2001 to 2003-archive sites are included)

Organization	2001	2002	2003	Archive sites (1996 to 1999)	Total sites
CFB	97	52	18	145	312
UCD	31	59	0	0	90
UCC	30	39	0	0	69
NUIG	16	15	0	0	31
UU	4	13	0	0	17
TOTAL	**178**	**178**	**18**	**145**	**519**

Table 2.5: Number of sites electrofished and used in the analysis, per stream order and Q-value

					Q-value			
Stream order	**1**	**2**	**2-3**	**3**	**3-4**	**4**	**4-5**	**5**
1	0	1	5	13	7	11	3	4
2	0	1	3	42	13	53	16	5
3	1	0	4	41	19	60	51	11
4	0	0	4	27	10	31	21	3
5	0	0	0	1	0	4	1	0
6	0	0	0	0	2	2	0	0
Totals (470)	**1**	**2**	**16**	**124**	**51**	**161**	**92**	**23**

2.4.1 Q-values versus altitude

Q-values were located over a range of altitudes, from 5.5m (Q2-3) to 262.8m (Q4) (Table 2.6). Contrary to the situation in many other European countries where generally speaking it is very difficult to find a high quality site in lowland areas, a number of the "high" quality sites surveyed during this project were located in lowland areas ("High" quality, minimum altitude= 8.2m). "Good" quality (Q4) sites ranged in altitude from 10.1m to 262.8, "moderate" quality (Q3-4) sites ranged from 4.2 to 174.7m, poor quality sites ranged from (5.5m to 203.2m). Statistical analysis showed that there was a significant increase in altitude from Q1 to Q5 (One-way ANOVA, df = 471, F=4.59, P=0.001) (Table 3.6). Further analysis using LSD post hoc tests showed that altitudes at "High" status sites were significantly higher than altitudes at "Poor" (P=0.001), "Moderate" (P=0.015) and "Good (P=0.007) ecological status sites.

Table 2.6: Q-values versus altitude (includes mean, standard deviation, standard error minimum, maximum and median altitude values for each Q-value class)

		Altitude (m)						
Q-value	**Ecological**	**Mean**	**SD**	**SE**	**Min**	**Max**	**Median**	**N**

	status							
Q1	Bad	-	-	-	65.7	65.7		1.0
Q2	Bad	61.7	4.7	3.3	58.3	65.0	61.7	2.0
Q2-3	Poor	55.0	21.5	5.4	5.5	84.9	53.9	16.0
Q3	Poor	59.8	34.2	3.1	6.2	203.2	58.2	124.0
Q3-4	Moderate	63.2	34.1	4.8	4.2	174.7	62.7	51.0
Q4	Good	71.6	48.7	3.8	10.1	262.8	66.4	161.0
Q4-5	High	81.6	49.8	5.1	8.2	219.7	75.6	95.0
Q5	High	105.6	56.8	11.8	27.8	214.3	83.6	23.0

2.4.2 Spatial variation on core rivers

Oona Water

All sites had low fish densities in both years (zero to 0.09 trout m^{-2}) and all four sites were moderately polluted (Q3) in 2001, three of the sites were similarly classified (Q3) in 2002 when the uppermost site (UU001) had improved (slightly polluted, Q3-4). Very low oxygen concentrations were recorded on 16th and 17th May 2002 when DO remained at or below 5.0 mg l^{-1} for 9 to 10 hours falling to 3.68 mg l^{-1} but no evidence of external pollution was detected.

Rye water

Water quality on the Rye Water fluctuated between Q3 and Q3-4 over the ten year study period. Brown trout fry densities were only available from 2000 to 2003 and these are presented in Table 2.7. Brown trout fry density was highest in 2000 and lowest in 2002. Salmon fry density was generally lower than trout fry density and the lowest levels were also recorded in 2002.

Table 2.7: Trout and salmon fry densities (including 95% C.I.) on the Rye Water from 2000 to 2003.

	Brown trout fry		Salmon fry	
Year	Site 1	Site 2	Site 1	Site 2
2000	0.12	0.09	0.03	0.02
2001	0.008	-	0.005	0.001
2002	0.001	0.001	0.001	0.001
2003	0.1 (0.016)	0.049 (0.007)	0.085 (0.016)	0.059 (0.024)

Additional archival data was obtained from UCD for the period 1994 to 2000. The density of (1+ and older) brown trout and salmon parr are presented in Figures 2.2a and 2.2b. Between 1994 and 2003, brown trout density varied from 0.05 to 0.31 m^{-2} and remained higher than salmon densities in both stretches. Trout density was highest in 1994 at Site 2. Thereafter, densities remained marginally higher in Site 2, but in 1995 densities were similar at both sites (Fig. 2.2a and 2.2b). Brown trout density was lowest in 1996, there was some improvement in each of the following years up to 1998 but it declined steadily thereafter.

Salmon parr densities remained low throughout the study period ranging from 0.001 to 0.12 m^{-2}. The highest salmon density was found in 1995 at site one.

Dunkellin River

A slight deterioration in quality was noted from 2001 to 2002, the mean overall Q-values decreased significantly ($p<0.05$.) from 3.0 (± 0.5) to 2.8 (± 0.4). This reflects a decline in quality from the 1970s & 1980s (Flanagan & Toner 1972; Flanagan & Larkin 1992).

Ten fish species were recorded in 2001 and 2002 (brook lamprey, salmon, brown trout, eel, stoneloach, pike, 3-spined stickleback, 10-spined stickleback, gudgeon, minnow. The overall pattern in species richness of the assemblages was shown to be inversely correlated with distance from the sea ($r_S=-0.67$; $p<0.01$) for the 1986 survey but not for 2001 or 2002, however, species richness of the site fish assemblages was positively correlated with stream order in all three surveys ($r_S=0.81$; 0.76 & 0.68 respectively for 1986, 2001 and 2002 surveys with $p<0.01$).

Robe River

Trout were the most commonly observed species followed by 3-spined stickleback, minnow and stoneloach. Fish species composition was calculated for each site surveyed and brown trout, particularly fry (0+ age group), were the dominant fish in 11 of the 14 tributary sites during 2001 (Fig. 2.3). Three-spine stickleback dominated stocks at two sites and minnow at one, indicating poor water quality at these three locations. Brown trout fry and parr (1+ age group) were the dominant fish groups at all sites during the 2002 survey, but three-spined stickleback composition was high (>20%) at five sites. Trout fry and parr were again dominant at ten sites in 2003 and three-spine stickleback dominated the fish community (>60%) at three sites, indicating a deterioration in water quality at these sites.

Water quality and salmonid densities fluctuated at most sites over the three years of the survey. Details of water quality (Q-value) and trout densities are presented below for a selection of sites for which additional information was also obtained from CFB archives (O'Grady *pers comm.*).

Water quality on the Claremorris golf club stream site was poor (Q2-3) in 2001 and a thick carpet of filamentous algae was present; no trout were recorded (Fig. 2.4a). Water quality improved to Q3 in 2002 and 2003, trout re-colonised the site and increased numerically in 2003. The abundance of filamentous algae had also decreased substantially in 2003. Only 3–spined stickleback occurred at this site in 2001, they were absent in 2002 and 2003.

No brown trout were recorded at the Scardaun stream site in 1996 (O'Grady unpublished data), due to poor water quality (Q2-3). Trout occurred in 2002 (1+ dominant) when water quality was good (Q4) and when quality declined in 2003 (Q3) trout density declined slightly and trout fry dominated the stock on this occasion (Fig. 2.4b).

Fry and parr densities were good on Mayfield stream (site 1) in 2001 (Q4) and 2002 (Q3-4); there were no trout fry present in 2003 (Q3) and trout parr densities, relatively consistent in 2001 & 2002, declined dramatically in 2003 (Fig. 2.4c). Trout fry densities decreased between 2001 and 2003 as did the Q-value (Q4 to Q3) at Mayfield site two, possibly due to siltation from land drainage works observed further upstream in January 2003 (Fig. 2.4d).

Slaney River

The fish stocks at each site in the Slaney catchment were surveyed five times between 1992 and 2003 (W. Roche, unpublished data) and the results are summarised in Appendix 5. The results presented are minimum estimates based on two run depletion samples in all cases (Crisp, *et al.*, 1974).

The fish stock at Eldon Bridge on the Slaney, was consistently dominated by juvenile salmon (Appendix 5). The Rahanahask Bridge site on the Clody tributary supported a more balanced stock of trout and salmon. Salmon fry (0+ age group) consistently dominate stocks on the Derreen river (Tinkers Bridge). The average densities of salmonids at these three clean survey sites (predominantly Q4-5 to Q5, occasionally Q4) were 0.559, 0.590 and 0.587 m^{-2} at Eldon Br., Rahanahask Br., and Tinkers Br., respectively (Fig. 2.5).

Since 1993 the water quality at the Boro river site has only ever been "good" (Q4) at best and frequently drops to doubtful to fair (Q3-4) with evidence of slight pollution common throughout its length. Normally trout are marginally more numerous at this site than salmon but salmon fry contributed to the highest stock density (1.592 m^{-2}) at this location in 1995 (Appendix 5). The river was surveyed by the EPA later that autumn and this section was classified as slightly polluted (Q3-4).

Juvenile salmon are normally more numerous than trout at the survey site on the River Bann. Fish stocks fluctuated greatly at this site, from low density (0.229 m^{-2}) in 1993 to a high density (1.36 m^{-2}) in 2000. Salmon fry dominated the stock in 2000 but trout fry were more abundant than salmon in 2003. Water quality here declined from good (Q4) in 1991 to doubtful (Q3) in 2001.

2.4.3 Relationship between fish and Q-values

Fish species richness in relation to Q-values and altitude

Overall a total of sixteen species of fish were recorded at the sites included in the dataset (Table 2.8). Brown trout were the most common fish species, followed by salmon, eel, 3-spined stickleback, stoneloach and minnow. Juvenile lampreys, although widely distributed, were only recorded at 14.6% of the sites. Sea trout, pike, roach, gudgeon, perch, ten-spined stickleback and flounder were recorded at a small number of the sites (Table 2.8).

Fish species richness was calculated for each Q-value group. The highest fish species richness was recorded at Q3-4 sites (mean = 3 species) because 'sensitive' and not so sensitive taxa co-exist at these locations (a similar finding was noted with macro-invertebrates (McGarrigle *pers comm.*)) (Fig. 2.6). The general trend for species richness in relation to Q-value was to increase from zero fish species at Q1 to a maximum diversity at Q3-4 and decrease slightly to Q5 (mean = 2 species) (Fig. 2.6). The maximum number of species recorded at any one site was eight (Q3 and Q3-4 sites). Statistical analysis showed that there was a significant difference in species richness between Q-values (One way ANOVA, df=510, F=4.854, P=0.001). LSD post-hoc tests also revealed that the mean number of species present at Q2 was significantly lower than the mean number of species at all other Q-value sites (LSD post hoc tests, Q2 v^s Q2-3 p=0.01, Q2-3 v^s Q3 p=0.0001, Q2 v^s Q3-4 P=0.0001, Q2 v^s Q4 P=0.0001, Q2 v^s Q4 p=0.0001 and Q2 v^s Q5 p=0.0001). LSD post hoc tests also showed that the mean number of species present at Q2-3 sites was also significantly lower than at four other Q-value site groupings (LSD post hoc test, Q2-3 v^s Q3 p=0.002, Q2-3 v^s Q3-4 p=0.003, Q2-3 v^s Q4 p=0.001 and Q2-3 v^s Q4-5 sites p=0.006 but not at Q5 sites (P=0.13).

In general fish species richness decreased with increasing altitude (Fig. 2.7). The highest values of species richness were recorded in the 0-10m and 11-50m altitude categories (mean 5.3 and 4.3 species respectively) and lowest species richness was recorded at sites greater than 100m altitude (mean 3.1, 3.3. 3.6 and 3.0). Only one species was recorded at the highest altitude site (UCD068-Vartry tributary site), i.e. brown trout (0+ and 1 + & older trout) (Table

2.8). Most species, apart from sea trout and flounder, were common at altitudes less than 100m. Eel, minnow, stoneloach, 3-spine stickleback, lamprey, brown trout and salmon were common at all sites less than 250m (Table 2.8). Coarse fish such as pike, perch, roach and other fish species such as 10-spine stickleback and gudgeon were absent or rare at altitudes over 150m (Table 2.8).

Table 2.8: Percentage of sites in each altitude class at which each fish species occurred (the number of sites surveyed in each class is also shown)

Species	% sites						
Altitude class (m)	**0-10**	**11-50**	**51-100**	**101-150**	**151-200**	**201-250**	**251-300**
Eel	17	32	46	33	37	50	0
Minnow	17	20	16	25	21	13	0
3-spine stickleback	50	34	41	45	32	25	0
10-spine stickleback	17	2	3	2	0	0	0
Stoneloach	0	24	24	24	37	25	0
Gudgeon	0	2	1	2	0	0	0
Pike	0	8	4	0	0	0	0
Perch	0	2	5	0	5	0	0
Roach	0	0	0	4	5	0	0
Flounder	0	0	1	0	0	0	0
Juvenile lamprey	0	12	15	20	16	25	0
Trout	83	86	91	87	100	100	100
Salmon	17	33	47	44	32	63	0
Sea trout	0	0	1	0	0	0	0
No. sites	**6**	**155**	**228**	**55**	**19**	**8**	**1**

Fish species composition in relation to Q-values

Fish species composition was calculated for each site at each Q-value (Figs. 2.8 and 2.9). Salmon and trout were treated separately and also combined as total salmonids, because of the absence of salmon at a number of sites due to the presence of impassable barriers downstream of the sites (Table 2.9 and Fig. 2.9). Two additional fish groups were also used, i.e. salmonid fry and salmonids 1+ and older. Seven fish species were found at less than 5% of sites (20 or less sites) (Table 2.9). Juvenile lampreys were combined into one fish group as the juveniles of the individual species were not identified in the field. The brown trout was the most widely distributed species, being found at 90% of the sites, followed by salmon 41.3%, eel 39.41%, and 3-spined stickleback 38.6%. Flounder were the least common, being present at only 2 sites (0.42%) (Table 2.8 and Fig. 2.8).

Results indicate that salmonids (trout and salmon) were the dominant species at Q3 to Q5 sites whereas 3-spined stickleback were the dominant fish species at the more polluted sites (i.e. Q1 to Q2-3) (Figs. 2.8 and 2.9). Statistical analysis (One-way ANOVA) showed that water quality, as indicated by Q-values, had a significant effect on the percentage composition (log transformed) of the fish community particularly, 3-spine stickleback, 10-spine stickleback, stoneloach and all salmonid groups.

Table 2.9: List of fish species recorded during the project (scientific and common names)

	Scientific names	Common names	Number of sites	% sites	Mean density (No./m^2)	Max density (No./m^2)	River where max density recorded
	Salmonidae						
1	*Salmo salar* (L.)	**Salmon (total)**	195	41.31	0.558	2.989	Feale, 2001
		0+ salmon	166	35.17	0.523	2.619	Owennalacken, Laune, 2001
		1+ & older salmon	167	35.38	0.413	0.985	Glendasan, Avoca, 2002
2	*Salmo trutta* (L.)	**Brown trout (total)**	423	89.70	0.507	3.429	Crover, Sheelin, May 1998
		0+ trout	377	79.87	0.484	3.377	Crover, Sheelin, 1998
		1+ & older trout	374	79.24	0.403	1	Lemonstown, Liffey, 2002
3		**Sea trout***	2	0.42	0.684	0.940	Camac, Liffey, 2002
	Esocidae						
4	*Esox lucius* (L.)	Pike	20	4.24	0.356	0.752	Cloonbur R., Mask, 1998
	Anguillidae						
5	*Anguilla anguilla* (L.)	Eel	186	39.41	0.353	0.985	Owenglin, 2002
	Cyprinidae						
6	*Rutilus rutilus* (L.)	Roach	4	0.85	0.267	0.330	Millpark R, Corrib, 1998
7	*Phoxinus phoxinus* (L.)	Minnow	87	18.43	0.428	1.500	Robe,2002
8	*Gobio gobio* (L.)	Gudgeon	6	1.27	0.364	0.522	Rafford, Dunkellin, 2001
	Petromyzonidae						
9	*Petromyzon marinus* (L.)	Sea lamprey					
	Lampetra fluviatilis (L.)	River lamprey	69	14.62	0.336	1.000	L. Coolin trib, Cloonbur, 1998
	Lampetra planeri (Bloch)	Brook lamprey	*(juveniles combined into one group)*				
	Percidae						
10	*Perca fluviatilis* (L.)	Perch	15	3.18		0.835	Carrowkerribly, Moy, 2002
	Gasterosteidae						
11	*Pungitius pungitius* (L.)	Ten-spined stickleback	11	2.33	0.402	0.792	Rafford, Dunkellin, 2002
12	*Gasterosteus aculeatus* (L.)	Three-spined stickleback	182	38.56	0.515	5.177	Kelystown, Rye Water, 2002
	Cobitidae						
13	*Barbatula barbatula (L.)*	Stoneloach	114	24.15	0.375	0.986	Brickens stream, Robe, 2002
	Pleuronectidae						
14	*Platichthys flesus* (Duncker)	Flounder	2	0.42	0.647	1.000	Delvin, Boyne, 2002

Note: Species shaded in red were not recorded at reference sites

Analysis showed that the percentage composition of 3-spined stickleback was highest at Q2-3 and decreased as water quality improved (Fig. 2.9). LSD post hoc tests divided 3-spined stickleback into four distinct groups, i.e. group 1 (Q2-3), group 2 (Q3), group 3 (Q3-4 and Q4) and group 4 (Q4-5 and Q5) (Fig. 2.9). In general, the percentage composition of all the salmonid groups increased in relation to water quality from Q2-3 to Q5. Further statistical analysis using LSD post hoc tests showed that two salmonids groups, i.e. total salmonids and salmonids (1+ & older) were the best indicators of water quality, in terms of species composition, as indicated by Q-values (Fig. 2.9). LSD post hoc tests divided total salmonids into four statistically different groups, i.e. group 1 (Q2-3), group 2 (Q3), group 3 (Q3-4 and Q4), and group 4 (Q4-5 and Q5) (Fig. 2.9). LSD post hoc tests divided salmonids (1+ & older) into five distinct groups, i.e. group 1 (Q2-3), group 2 (Q3), group 3 (Q3-4 and Q4), group 4 (Q4-5) and group 5 (Q5) (Fig. 2.9). Figure 2.10 illustrates the relationship between two indicator groups (non-salmonids and total salmonids) over a water quality gradient as measured by Q-values. This indicates that non-salmonid fish species dominate the fish community at Q2-3 sites and gradually decrease to less than ten per cent of the fish population at Q4-5 and Q5 sites.

Fish species abundance in relation to Q-values

Fish species abundance as indicated by minimum densities (no.m^{-2}) was calculated for each site at each Q-value (Fig. 2.11). One-way Analysis of Variance showed that water quality as indicated by Q-values had a significant effect on the abundance of certain fish species particularly, 3-spined stickleback and the seven salmonid groups. The abundance of three-spined stickleback decreased with increasing water quality, whereas the abundance of salmonids, in general, increased with increasing water quality. LSD post tests indicated that the abundance of 3-spined stickleback (log transformed) was significantly higher at sites with a Q-value of Q2-3 and Q3 than sites with higher Q-values (Q3-4 to Q5) (Fig. 2.11). Statistical analysis also showed that the abundance (no m^{-2}) of the salmonids 1+ and older group was the best indicator of water quality (in terms of fish abundance). Abundance of salmonids 1+ and older was significantly higher at Q4-5 and Q5 than at any other sites (Fig. 2.11). LSD post hoc tests divided this taxonomic group into three significantly different groups, i.e. group 1 (Q2-3 and Q3), group 2 (Q3-4 and Q4) and group 3 (Q4-5 and Q5) (Fig. 2.11). However, LSD post-hoc test also showed that 1+ and older salmon were significantly different between sites rated moderate (Q3-4) and good (Q4) (Fig. 2.11).

2.4.4 Classification of fish using TWINSPAN

Removal of rare species (gudgeon, roach, flounder and sea trout were present at 4, 3, 2 sites respectively and sea trout were only present at 1 site) resulted in 12 taxonomic groups being used in the analysis. TWINSPAN was then carried out on the remaining 385 sites (quantitative data) and produced 14 site groupings. However, a TWINSPAN division was only accepted if the groups differed significantly ($P<0.05$) according to MRPP (PC-ORD) and if there were more than 5 sites present in the group and this resulted in 14 final fish groups (Fig. 2.12). The first level of the TWINSPAN hierarchy separated the 385 sites into two groups of 357 and 28 sites. Of the 28 sites on the right hand side of the TWINSPAN tree differed due to the presence of high densities of 3-spined stickleback. The second level divided the 357 sites into 172 sites and 185 sites. The 185 site grouping differed due to the presence of salmon fry and salmon 1+ and older. The third level divided the 172 and 185 site groupings into four further groups, the right hand side of the TWINSPAN tree further subdivided by the presence of salmon (Fig. 2.12).

2.4.5 Classification of macroinvertebrates using TWINSPAN

Removal of rare taxa resulted in 62 taxonomic groups being used in the TWINSPAN classification. TWINSPAN was then carried out on 391 sites (using abundance data) and produced 237 site groupings. After MRPP analysis 37 final macroinvertebrate groupings remained (Fig.2.13).

The first level of the TWINSPAN macroinvertebrate hierarchy separated the 391 sites into two groups of 333 (moderate to good sites with Q-values ranging from Q3 to Q5) and 58 sites (moderate to poor sites with Q-values ranging from Q1, Q2, Q2-3 and Q3) (Fig. 3.13). The group of 58 sites were further divided into two groups (23 and 32 sites) in level 2 due to the presence of Tubificidae (Fig. 2.13). The 333 sites were divided into two groups, one of 180 (due to the presence of *Ecdyonurus* sp., Nemouridae and Glossosomatidae) and one of 153 sites (due to the presence of *Gammarus* sp.). There were six further divisions in the TWINSPAN analysis resulting in 62 final macroinvertebrate groupings. In general the grouping of the macroinvertebrate sites reflected the Q-value gradient as the majority of clean sites (Q4-5 and Q5) were split from the majority of sites which were enriched in the first division of the dendrogram (Fig. 2.13). However the separation was not as clear for sites rated Q3 to Q3-4 (Fig. 2.13).

2.4.6 Concordance of fish and macroinvertebrates

It was possible to test for concordance between the fish and macroinvertebrate communities once the TWINSPAN groups were derived. The 37 macroinvertebrate groupings were imposed on the fish abundance dataset and the 14 fish groupings were imposed on the macroinvertebrate data using ANOSIM in order to determine whether the classification for one group would apply to the other. The highest similarity is given a rank of 1 and ranges from +1 to -1. Positive values indicate differences among groups (McCune and Grace, 2002). A value of 0 occurs if the high and low similarities are perfectly mixed and bear no relationship to the group. When R is significant there is evidence that the samples within groups are more similar than would be expected by random chance (Seaby *et al*, 2004).

Macroinvertebrate TWINSPAN end groupings were statistically significant (R=0.633, p=0.001) (Table 2.10). There was also a statistically significant separation of groups when these groupings were imposed on the fish data, although the R value was low (R=0.238, P=0.001) (Table 2.10). Likewise, there was a statistically significant separation when the fish groupings were applied to the macroinvertebrate data (R=0.065, P=0.001). ANOSIM was also used to investigate how well Q-values separated both biotic datasets. The Q-values separated the macroinvertebrates to some degree, however, they did not significantly separate the fish groups (Table 2.10), indicating that other factors are structuring the fish communities and that the impact of water quality acts on top of these.

Table 2.10: Concordance (cross tests using ANOSIM) between fish and macroinvertebrate datasets (significance level (%) = 0.1).

	Validated groupings			
Dataset	**Fish**	**P**	**Macroinvertebrates**	**P**
Fish	0.261	0.001	0.238	0.001
Macroinvertebrates	0.065	0.006	0.633	0.001
Q-value	-0.013	0.721 (ns)	0.095	0.001

2.4.7 Mantel correlations

A Mantel test was carried out as a final measure of concordance between the fish and macroinvertebrate communities. The Mantel test evaluates the relationship between two similarity matrices and r ranges from -1 to +1 (McCune and Mefford, 1999). The Mantel test produced an r value which is statistically significant of 0.197 (P<0.001, t=8.8684). This also indicates that there is a positive association between the fish and macroinvertebrate matrices.

2.4.8 Environmental definition of macroinvertebrate and fish sites using ordination

Detrended correspondence analysis (DCA) was carried out on the macroinvertebrate and fish data to assess the gradient length of the first DCA axis (in SD units). The gradient lengths for fish and macroinvertebrates were 2.527 and 3.24 respectively. These gradient lengths suggested that methods based on a unimodal response model (i.e. Canonical Correspondence Analysis) were more suited to the data than linear models.

The pattern of variation in fish and macroinvertebrate community composition at 390 sites in relation to 28 environmental parameters was analysed using Canonical correspondence analysis (Figs. 2.14 and 2.15 respectively). Both biotic groups were subjected to an identical environmental matrix. Table 3.11 gives a summary of the CCA analysis for fish and macroinvertebrates using forward selection of variables.

Analysis of the fish data in relation to the environmental variables indicated that alkalinity, barrier downstream and dimensions of the site surveyed (represented by surface area) were the three most important variables influencing the fish and community composition. Similarly, alkalinity, northing, barrier downstream and geology were found to be the most important variables influencing the macroinvertebrates. This indicates that most of the variation in fish and macroinvertebrate composition is caused by the environmental variables (physical) (Table 2.11). The first canonical axis for fish accounts for 14% of the variation, however the first axis only accounts for 4.6% of the variation in macroinvertebrate composition. A Monte Carlo test was used to test for significance of the canonical axes and indicated that the amount of variability explained by the environmental variables was significant for the first and second axes (P=0.001).

Table 2.11: Summary of Canonical Correspondence Analysis using forward selection of variables. The significance of all axes was P=0.001.

	Axis 1	Axis 2
	Fish	
Eigenvalue	0.4096	0.2130
Cumulative percentage variance	14.1052	21.44
Multiple species/environment scores	0.7395	0.6120
	Macroinvertebrates	
Eigenvalue	0.1333	0.0129
Cumulative percentage variance	4.634	6.289
Multiple species/environment scores	0.7976	0.5954

Note:

- The eigenvalue represents the importance of each axis and how well the environmental variables explain the species data ranging from 0 to 1.
- The cumulative percentage variance represents how much the variables derived from the macroinvertebrate data are explained by the linear combination of environmental variables.
- The multiple species/environment correlation shows the strength of the relationship between macroinvertebrates and environmental parameters (a value close to 1 indicates that the environmental variables are having an appreciable effect).

Figure 2.14a shows that fish species, such as 3-spined stickleback, indicative of enriched sites are located on the right hand side of the biplot associated with silt and alkalinity. Coarse fish species such as pike and perch are associated with percentage pool. Juvenile salmon were associated with mean wetted width and stream order. Juvenile salmon were negatively correlated with the existence of a barrier downstream of the site. In general many of the more enriched sites (Q2-3 and Q3) from the Robe and Liffey catchments are located to the right of the biplot, whereas cleaner sites Q4 and Q4-5 are located to the left of the biplot (Fig. 2.14b). Trout were associated with percentage pool, northing and barrier downstream. Alkalinity was the single biggest variable found to be influencing the fish population, indicating that fish abundances increase with increasing alkalinity (Fig. 2.14a).

Alkalinity was also found to be the single biggest variable influencing the macroinvertebrate population (Figs. 2.15a and b). Figures 2.15a and 2.15b show that in general the macroinvertebrate TWINSPAN groups are situated at either ends of the main environmental gradients present in the dataset. Sites indicating poor water quality (low Q-values) are situated on the left hand side of the biplot, and were associated with variables such as percentage mud and silt, alkalinity, percentage instream cover and percentage glide. Whereas the high quality sites (Q4-5 and Q5) are situated on the right hand side of the biplot and were associated with mean wetted width, percentage boulder, stream order, percentage riffle and altitude (Fig. 2.15a). Intolerant macroinvertebrate taxa, such as Chloroperlidae, Heptageniidae, Perlodidae, and Perlidae were situated on the right hand side of the biplot. Tolerant taxa such as *Asellus* sp., *Chironomus* sp. and many mollusc species were located on the left hand side of the biplot (Fig. 2.15b).

2.5 Discussion

A principal objective of this project was to investigate biotic responses to environmental pressures specifically in the context of EPA Water Quality Ratings (Q-value) and fish. The specific aim of this workpackage, was to assess the impact of water quality, as evidenced by the EPA's Quality Rating System on riverine fish stocks, to establish if a relationship exists between fish and varying Q-values (Q1 to Q5) and to assess the feasibility of using fish assemblages as biological indicators of ecological quality in Irish rivers. The EPA Q-value system has been shown to be a robust indicator of lotic water quality and has been linked with both chemical status and land use pressures in catchments (Clabby *et al*., 1992; McGarrigle, 1998).

Fish communities at over 500 locations in 1^{st} to 6^{th} order streams were analysed in this study. With the exception of flounder, which only occurred at 1% of sites, native species were well distributed at all elevations 0-263m, trout were the most widespread, occurring at 83 to 100%. Non-native (e.g. coarse fish) species exhibit a sporadic distribution in Irish rivers being absent from many catchments. Of these species pike and perch were each encountered at 12% of sites overall (11 to 100m), gudgeon (6 sites) occurred at 5% (11 to 150m) and roach although expanding geographically only occurred at 1% of sites (101 to 200m).

In this study stocks were only assessed in wadable stretches or where depth was <1.5m (boat electric fishing). Consequently investigations were mostly conducted in small to medium rivers (stream order 1 to 4 accounted for 460 sites) across the full range of Q values at elevations ranging from 4.2m (Q3-4) to 263m (Q4). Only ten locations were surveyed on larger rivers of which six sites were stream order 5 (Q value 3-4 to 4-5) and four were stream order 6 (Q value 3-4 to 4). Fish stock surveys were conducted over a wide range of altitudes. The study showed that high quality (Q4-5 and Q5) sites surveyed during the project were more common at high altitudes than poor quality sites, however, a number of these high quality sites were located in lowland areas. This is contrary to the situation in many European countries where generally speaking it is very difficult to find unpolluted sites in lowland areas. There are a number of relatively large catchments in Ireland of at least "good" status and indeed some with "high" status extending to their lower reaches, e.g. sections of the lower river Moy (Donohue *et al*., 2006).

In the rivers of Western Europe, Huet, (1959), identified four main – and usually quite distinct - biological zones each of which has a characteristic fish fauna with a diagnostic or "key" species. The typical zonal sequence is (1) trout, (2) grayling (*Thymallus thymallus*), (3) barbel (*Barbus barbus*) and (4) bream, from headwaters to the sea (Welcomme 1985). Cyprinid species are most common in the latter two zones with moderate current and flow. This longitudinal pattern based on four zones, provides a useful summary of the habitat requirements of European freshwater fishes (Cowx and Welcomme, 1998). Huet (1959) advises that the physical characteristics of the trout zone differ in physically distinct stretches, from the headwaters to the coastal plains, "but the water is always cool and well oxygenated". Huet (1959) also states that the four icthyological zones really represent two faunistic regions: an upper salmonid region of cooler waters (trout and grayling zones) and a lower cyprinid region of warmer waters (barbel and bream zones).

In the bream zone current is slight, summer water temperature high, dissolved oxygen often fairly low; the water is often turbid and the depth may exceed 2.0m. The associated fish species in this zone are carp, tench, roach, pike, perch and eels. Huet (1959) also suggests that in a given "bio-geographical area, rivers or stretches of rivers of like breadth, depth and slope have nearly identical biological characteristics and very similar fish populations". The fish zonation is mostly the result of the physical characteristics stream gradient, stream width, current speed and temperature.

Bream have been introduced but are not widespread, with the exception of the River Shannon and they are confined to the very lower deeper reaches of the few rivers in which they occur. Grayling and barbel do not occur in Ireland. The common bream is a fish of lowland rivers and prefers rich, muddy, weedy areas, reservoirs and slow-flowing rivers (Maitland and Campbell, 1992). In general Ireland doesn't have the very large, silt laden rivers that are common in the plains of continental Europe where tillage rather than grassland is the dominant form of land use and inland population density tends to be much higher than in Ireland (McGarrigle, 2001). Roach (the zoogeography of which has been extended in Ireland by anglers) are capable of colonising areas of faster flow but were only encountered at four locations in this study. Consequently, the fish community structure characteristic of continental rivers does not occur in the vast majority of Irish rivers especially in those systems where cyprinids have not been introduced.

While the physical characteristics which form the basis of Huet's (1959) fish zonation exist (to a limited extent) only two of the 'key' indicator species, trout and bream, currently occur in Ireland. This ERTDI funded study, for logistical reasons is confined almost exclusively to shallow wadable waters and in the absence of pollution, brown trout and, where access and water quality permits, salmon normally dominate such habitats. Bream zones have not been surveyed. In Ireland the geographical distribution of coarse fish species, though patchy is expanding and gudgeon, minnow, stone loach, roach, dace and bream, (recently live chub (*Leuciscus cephalus*) have been confirmed at one location) may eventually populate more of these locations. In Irish rivers current speed, oxygen content and water temperature favour salmonids and these factors may militate against widespread colonisation by coarse fish. However, the temperature regime, river flows and thus current velocity are expected to change, if climate change predictions are realised (Sweeney, *et al*, 2002). Elevated water temperature, (influencing growth and reproduction) and reduced flows could favour cyprinids over salmonids in Irish midland and eastern rivers particularly.

Physical (hydromorphological) factors primarily determine the distribution of fish species and community composition along a river corridor from high to low gradient (Huet, 1959; Welcomme, 1985 and Cowx and Welcomme, 1998). This is also indicated by the correlation of fish in Chapter 3 with distance from source, wetted area, stream order, etc. Ordination analyses also showed sticklebacks to be associated with silt/mud and alkalinity (features of lower elevations) and salmon with mean wetted width, stream order and distance from source. Nonetheless biota (including fish) will change due to external pressures independently of hydromorphological factors. This is apparent from the results obtained at specific locations in the core rivers (e.g. the River Robe sites and the River Slaney) over a prolonged period. This change in biological communities in response to external environmental pressure (physico-chemical) is demonstrated in sequential national reports of water quality published by the EPA over the past 30 years (Clabby *et al*., 2001 and 2002; Flanagan and Toner, 1972). The extent of high quality (Q5 and Q4-5) channel has declined and a three to five-fold increase in moderately and slightly polluted stretches occurred between 1971 and 1996 (Bowman and Clabby, 1998).

In the EPA Quality Rating scheme high quality is indicated by Q4-5 and Q5. Whilst some impairment is evident at Q4, the ecological conditions at such locations are considered to be acceptable to salmonids. However, this is an assumed relationship based on visual observations and an important element of the project was to research the validity of this hypothesis. The present study found that fish community composition (% composition) did change with increasing Q-value and it was statistically possible to separate some fish groups in relation to a number of Q-value groups. In terms of species percentage composition two salmonids groups, i.e. total salmonids and salmonids (1+ & older) were identified as the best indicators of water quality. Total salmonids divided into four statistically different groups, i.e. group 1 (Q2-3), group 2 (Q3), group 3 (Q3-4 and Q4), and group 4 (Q4-5 and Q5). Salmonids (1+ & older) divided into five distinct groups, i.e. group 1 (Q2-3), group 2 (Q3), group 3 (Q3-4 and Q4), group 4 (Q4-5) and group 5 (Q5).

In terms of fish abundance (no. fish m^{-2}) the salmonids 1+ and older group was the best indicator of water quality. Abundance of salmonids 1+ and older was significantly higher at Q4-5 and Q5 than at any other sites. Further statistical analysis divided 1+ and older salmonids into three significantly different groups, i.e. group 1 (Q2-3 and Q3), group 2 (Q3-4 and Q4) and group 3 (Q4-5 and Q5). Statistical analysis also showed that the abundance of 1+

and older salmon were significantly different between sites rated moderate (Q3-4) and good (Q4) quality.

This study shows a continuous decrease in salmonid abundance in relation to decreasing eological quality, however the change in salmonid abundance occurred at a slightly lower Q-value than was hypothesised i.e. there was a more drastic loss of salmonids at or below Q3 instead of below Q4 which was hypothesised. At locations classified as Q3-4 or Q3 conditions may be exacerbated in the warm summer months but may return to normal throughout the remainder of the year. Fish were absent from bad quality sites (Q2 or less). Nutrient enriched/organically polluted poor quality (Q2-3) sites were characterised by a high abundance of three-spined stickleback and no or very occasionally, very low numbers of salmonids. Trout and salmon occurred at enriched poor quality (Q3 sites) through to unpolluted high quality sites (Q5). The most significant fish community change occurred between Q2-3 and Q3 sites. At Q-values less than Q3, 3-spined sticklebacks dominated and at sites with Q-values in the range Q3 to Q5 salmonids were the dominant group.

Salmonids are generally more sensitive to low dissolved oxygen than cyprinid fish (Larkin and Northcote, 1969; Doudoroff and Shumway, 1970). Salmonids though present at enriched poor quality locations, are at risk and may be unsustainable in the longer term due to occasional pressures associated with the additive effect of high temperature and low oxygen, as prolific growths of benthic macro algae are characteristic features at such quality ratings (McGarrigle, 2001). A low number of salmonids were present in a few of the Q2-3 sites. In these instances they may have colonised from cleaner, neighbouring habitats or may have survived in clean water entering from side streams. The 3-spined stickleback appears to be relatively pollution tolerant (McCarthy & Kennedy, 1965; Maitland & Campbell, 1992) and is also a good coloniser of rivers recovering from severe pollution (Turnpenny and Williams, 1981). The absence of fish from the aquatic environment can be far more informative than their presence when assessing biological integrity.

The general trend for species richness in relation to Q-value was to increase from zero fish species at bad quality sites (Q1) to a maximum diversity at moderate quality (Q3-4) sites (mean = 3 species) and a slight decrease at high quality (Q5) sites (mean = 2 species). The maximum number of species recorded at any one site was eight (Q3 and Q3-4 sites). Species richness was highest at the eutrophic river sites (i.e. Q3-4) and included species such as trout, salmon, 3-spined stickleback, lamprey, stoneloach, minnow, roach and eels. Some of these species are more tolerant to adverse conditions than salmonids. For example, stoneloach are more tolerant of low oxygen concentration than trout (Elliott *et al*., 1994; Ekloev *et al*., 1999). Eels are also tolerant of low oxygen levels and can breathe air if necessary, but they usually prefer well oxygenated water (Thiel, *et al*., 1995). Although eutrophic stretches have their associated problems, they also offer a number of advantages to some fish species. Eutrophic streams have increased plant growth, which provides a better habitat for some fish species. In a study of a Danish lowland stream, vegetation changes with increasing eutrophication towards species with a capacity to form a canopy providing refuges from predation and a spawning substrate for species such as minnow, 3-spined stickleback and some coarse fish. According to Vannote *et al*. (1980) the abundance of fish and macroinvertebrates is generally highest at intermediate nutrient levels. Miltner and Rankin (1998) suggest that nutrient enrichment and overall water quality, barring gross pollution, will have an influence on fish communities in small streams. They found that the relative abundance of tolerant and omnivorous fish increased significantly in relation to nutrient enrichment (high concentrations of total inorganic nitrogen and total phosphorus) in headwaters, wadable streams and small rivers.

A low species richness was recorded at high quality Q4-5 and Q5 sites in this study, some of which were located upstream of natural barriers. McDonnell (2005) found that species richness was related to stream order and altitude and stated that "species richness" may not be a good indicator of water quality in Irish streams due to the paucity of fish species present. Low species number reflects the natural or reference status of the fish community in Ireland. Angermeier and Davideanu (2004) created a fish based Index of Biotic Integrity for assessing stream quality in Romanian streams, but left out the metric for total number of species and concluded that it was less related to water quality in depauperate fish assemblages. Many studies have reported changes in river fish community assemblage from sensitive species (e.g. salmonid species) to more tolerant species such as cyprinids associated with a decrease in water quality (Ekloev *et al*., 1999). The change in fish community structure is thought to be caused by a combination of effects including a decrease in dissolved oxygen and siltation of spawning gravel (Lee and Jones, 1991). Where cyprinids have not been introduced or colonised naturally, as is the case in most Irish river channels, clear distinctions are likely to be less evident. There is not a linear relationship between species richness and water quality in rivers in Ireland and this is not surprising in the circumstances, however, there is a relationship. At high quality sites only a few salmonid species are found, at heavily polluted sites no fish or only a few non-salmonid species occur, while at the enriched intermediate sites maximum diversity is found with a mix of tolerant and intolerant fish species. At these latter sites salmonids though present are possibly unsustainable and particularly at risk in warm summers due to severe fluctuation in oxygen at such locations (McGarrigle, 2001). Severe diurnal fluctuations in oxygen recorded in this study on the Rye Water and Oona Water appear to support this view.

The Oona Water was moderately polluted throughout its length and the fish densities and species richness were so low as to be of little value in the context of the spatial and temporal variation study. Dramatic diurnal variation in oxygen content is a significant feature of this channel and consistent with the oxygen regime often associated with Q-Values of Q3 and Q3-4. There was no clear relationship between salmonid densities (which varied relatively little) and water quality on the Rye Water, which fluctuated only slightly between Q3 and Q3-4 over the ten year study period. This is not unexpected as the range of variation in quality and densities is relatively restricted – typically a wide dynamic range is needed to show such relationships – both the fish densities and Q-Values were relatively static over the period. The results of the Dunkellin surveys and earlier data provide a baseline against which to evaluate future changes. However, as with the Rye Water, the Dunkellin has a relatively restricted range of water qualities over the years again making it difficult to generalise from this system.

Trout densities on the Robe varied within and between sites and, with the exception of few or no trout at poor quality (Q2-3) sites, these variations appear to be loosely linked to water quality. Stickleback numbers increased and declined at different sites over the three years of the study; high numbers of this species did appear to be linked with impairment in water quality at the Robe sites and trout did colonise sections when water quality improved. Biological assessment of water quality was not conducted at the same locations as the fish stock survey on the Slaney system (only at two of the five sites) nor were these surveys conducted concurrently (only once, in 1995, were fish stocks and biological assessment carried out in the same year). However, densities of total salmonids appear to be more stable at the better sites (Q4-5 & Q5) on the Derreen, Upper Slaney and Clody Rivers. In contrast, in the Bann and Boro Rivers, where water quality is mostly less than good status (Q3-4 to Q4), fish stock densities oscillate more widely attributable mainly to pulses in salmon fry (0+ age group) abundance. It does appear that there is not a strong relationship between densities of total salmonids and EPA water quality ratings in the Slaney dataset Elliott (1994) warns of the inadequacies of short-term studies to evaluate natural variability in fish communities and advises long term investigation to establish a baseline against which to measure change. This advice is substantiated by the results of the Slaney monitoring programme where at the Bann site the highest annual density is six times the lowest value and five times the lowest at the Boro site. The average value for total salmonids at all five sites over the ten year study period is 0.6 salmonids m^{-2}. Year to year variation appears not be related to water quality but to physical habitat and spawning effort. The River Slaney is an important river system for salmon with a tradition for high numbers of multi-sea winter fish. However, in recent years the estimated spawning effort is only a percentage of the conservation limit for this system (National Salmon Commission, 2006). It is unlikely, therefore, that density dependent factors currently limit the juvenile salmon in this system. High densities of fry at the R. Bann site in 1995 and the R. Boro site in 2000 were thought to be a result of improved adult escapement/ova deposition in the previous winter in both cases. The two locations with the highest stock density of salmon fry and the widest temporal range are situated on the tributaries where water quality is impaired.

The results from the present study show that there is concordance between fish and macroinvertebrate community composition in streams across Ireland. Kilgour and Barton (1999) examined the concordance between stream fish and benthos across environmental gradients in sourthern Ontario, Canada. They found a significant association between fish and macroinverterbates as they were both responding to similar environmental variables, i.e. temperature and stream size. In this study, both biotic groups were responding largely to similar environmental gradients as shown by CCA, these included alkalinity, the occurrence of barriers downstream and the latitude of the site. Concordance between the two biotic groups suggests that it may be feasible to use macroinvertebrate community structure to assess fish community composition across a water pollution gradient. Although there is concordance between the two groups there is also considerable overlap in the fish species abundance found at sites rated Q3 and above.

Fish community structure was related to alkalinity, the occurrence of barriers downstream, surface area of the site and wetted width. Pollution tolerant fish species, such as the 3-spined stickleback, occurred to the right of the bi-plot (Fig. 3.14) and were best described by alkalinity and mud and silt. Salmon were grouped to the left of the bi-plot and were associated with mean wetted width and stream order. Juvenile salmon were negatively correlated with the occurrence of barriers downstream. The productivity of fish populations has been linked to the productivity of benthic communities in rivers (Krueger and Waters, 1983). Fish occur at a number of trophic levels and in a number of trophic guilds and can also control macroinvertebrate structure through predation. Productivity is also linked to alkalinity (Lobon-Cervia and Fitzmaurice, 1988).

Analysis showed that tolerant macroinvertebrate taxa increased along a nutrient gradient at the expense of sensitive taxa. The CCA segregated eutrophic sites from clean sites and established that alkalinity, barrier downstream, latitude, geology, percentage glide and percentage instream cover were the main habitat variables which explained macroinverterbate community composition. The effects of nutrient enrichment on macroinvertebrates are well documented (Mason, 1996; Parr and Mason, 2003). Changes in species composition with decreasing Q-values were evident in this study. McDonnell (2005) found that diversity and evenness of macroinvertebrates was reduced at eutrophic sites and found that width, altitude, instream vegetation, velocity and percentage silt were the main variables influencing community composition. Cole (1973) found that aquatic insects, normally found in well-oxygenated riffle habitats, were rare in enriched stretches.

According to the CCA analysis there were a number of environmental (physical) variables which uniquely affected each biotic group. In addition to the variables listed above, fish were also related to stream order, distance from source and percentage pool, whereas macroinvertebrates were associated with geology, percentage glide, instream cover, altitude and percentage boulder. This difference may explain the low similarity (ANOSIM) and Mantel values (low R values) and the large degree of divergence between the biotic groups.

Although there is concordance between the two groups there is also considerable overlap in fish species abundance and composition found at sites Q3 and above. Results indicate that the fish abundance metric (i.e. number of fish m^{-2}) does not separate sites according to water quality as well as the alternate metric, i.e. fish species composition. Significantly the findings indicate that abundance of salmon (1+ and older) could be used as an indicator to separate "moderate" and "good" quality sites. However, this criterion can only be applied downstream of impassable barriers.

There is not a linear relationship between fish species richness and water quality in rivers in Ireland, however there is a relationship. According to McDonnell (2005) this should not be viewed negatively because disparate assemblages may provide a more complete view of the many human impacts on the aquatic ecosystem as one group may be more vulnerable to certain stresses than another. At high quality sites only a few salmonid species are found, at heavily polluted sites only a few or no non-salmonid species are found, while at the enriched intermediate sites maximum diversity is found with a mix of tolerant and intolerant fish species. Therefore, a simple metric of fish composition which combines species richness with a weighting of species for their tolerances may show a clearer relationship between Q-value groupings.

Kilgour and Barton (1999) state that it should be possible to use surveys of fish and benthos to diagnose the nature of disturbances (impacts) and macroinvertebrates respond to environmental cues much more quickly than fish. Karr (1981) pioneered the use of fish communities in an Index of Biological Integrity (IBI), but the low number of fish species and trophic guilds present in Ireland may present a limit to its use here. However, the IBI index has been adapted for use in many countries throughout the world including countries with low species richness (Harris and Silveira, 1999; Oberdorff *et al*., 2001; FAME CONSORTIUM, 2004; Joy and Death, 2004; Pont *et al*., 2006).

Twelve European countries participated in a project to develop and validate a common multimetric fish based index applicable to all European rivers (Pont *et al*., 2006). The project tested several fish-based assessment methods for the ecological status of rivers and the European Fish Index (EFI) was selected as the method most suitable to meet the requirements of the WFD (FAME CONSORTIUM, 2004). The EFI is a multi-metric predictive model that derives reference conditions for individual sites and quantifies the deviation between predicted and observed conditions of the fish fauna. The ecological status is expressed as an index ranging from 1 (high ecological status) to 0 (bad ecological status) (FAME CONSORTIUM, 2004). The EFI uses data from single pass electric fishing and employs 10 metrics based on species richness and densities. Ten environmental variables (nine variables account for local variability, e.g. altitude and slope and one variable, i.e. river region, is used to explain regional differences) and three sampling variables pertaining to the specific site and sampling strategy are used to predict reference values (FAME CONSORTIUM, 2004). This index may be transferable between catchments at the European scale (Pont *et al*., 2006). A number of limitations with the index have been identified for particular environmental situations, such as the outlet of lakes, predominantly spring fed lowland rivers, undisturbed rivers with naturally low fish density and heavily disturbed sites where fish are nearly extinct (FAME CONSORTIUM, 2004). Because numerous Irish rivers are relatively undisturbed and have naturally low fish diversity and density the EFI has yet to be calibrated and tested using the dataset derived for this project to investigate if it is applicable to Irish rivers.

2.6 Conclusions

Whilst the original objectives of the project do not specifically include the WFD the project nonetheless provides certain outputs which are useful in the context of the Directive, e.g. classification systems and a comprehensive GIS dataset for fish and other priority elements in wadable rivers.

The findings from the present study confirm that there is a relationship between fish community composition and Q-values. Fish community structure did change with increasing ecological status (from bad to high). Non-salmonid fish species dominate the fish community at “poor” quality (Q2-3) sites and gradually decrease to less than ten per cent of the fish population at “high” quality (Q4-5 and Q5) sites, whereas salmonids dominate the fish community at high quality sites and decrease to less than 20% at “poor” quality sites (Q2-3). The study also confirms that fish and macroinvertebrates are significantly associated in streams in Ireland. This project has produced a comprehensive and valuable data set for wadable rivers stream order 1 to stream order 4. These data have been collected using a standard format, they may be re-analysed at any future date and they will be used to test the EFI.

For WFD purposes metrics that cover the three main descriptors of fish populations are needed, i.e. “*Composition, abundance and age structure of fish fauna*”. In this study statistical analyses has shown significant differences between quality ratings and ecological class boundaries – four simple metrics have been identified in this study which can be used to separate the “High/good” and “Good/Moderate” boundaries for the WFD for fish (particularly for wadable river sites). Separation of the Good/Moderate boundary i.e. Q4/Q3-4 is particularly important but it is a relatively subtle change indicated by 1+ and older salmon which is only applicable to locations downstream of impassable barriers. These metrics are:

1. Percentage composition of total salmonids
2. Percentage composition of salmonids 1+ and older
3. Abundance of salmonids 1+ and older
4. Abundance of 1+ and older salmon (applicable to sites downstream of impassable barriers).

2.7 Recommendations

Further work should include

- Development of an index of biotic integrity for Irish rivers using the current dataset
- Assessment of fish community structure in deep sections of Irish rivers.
- Testing of the FAME (EFI) software using the current dataset to establish if the index can be successfully applied to the Irish fish community.
- A potential fifth metric has also been identified, i.e. species richness weighted for species tolerances could show a clear relationship between Q-value groupings. This requires further investigation.