Environmental RTDI Programme 2000–2006

Investigation of the Relationship between Fish Stocks, Ecological Quality Ratings (Q-Values), Environmental Factors and Degree of Eutrophication

(2000-MS-4-M1)

Synthesis Report

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Prepared for the Environmental Protection Agency

by

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WATER QUALITY

The Water Quality Section of the Environmental RTDI programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in this area. The reports in this series are intended as contributions to the necessary debate on water quality and the environment.

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Executive Summary

In Ireland the water quality of streams and rivers has been assessed using macroinvertebrates, chemistry and macroflora since the 1960s. The Water Framework Directive (EC Directive 2000/60/EC) (WFD) lists fish amongst the biological elements which should be used for the classification of ecological status of surface waters (rivers, lakes and transitional waters [estuaries]). This project was supported under the Environmental Protection Agency (EPA) ERTDI (2000–2006) programme to: (i) assess the impact of water quality, as evidenced by the EPA Quality Rating System (Q-values) on riverine fish stocks; (ii) assess the feasibility of using fish assemblages as indicators of ecological quality; and (iii) develop a predictive model which would have application in the context of the WFD. Investigation of specific questions regarding eutrophication pressures was also required. The project was awarded to and executed by an alliance of state agencies and academic institutions north and south of the border.

Using Comité Européen de Normalisation (CEN)-compliant electric fishing methods, a comprehensive dataset of fish and habitat variables was generated at 374 river locations, mostly in wadable 1st to 4th order streams, across the full range of EPA water-quality ratings. Archival material for fish and habitat variables was sourced for another 145 sites. Established EPA protocols were used to assess water quality at each location surveyed.

The study established that there is a relationship between fish-community composition and Q-values. Non-salmonids dominate the fish community at 'poor'-quality (Q2–3) sites but decrease to <10% of the fish population at 'high'-quality (Q4–5 and Q5) sites, whereas salmonids dominate the community at high-quality sites and decrease to <20% at poor-quality sites. It was statistically possible to separate a number of fish groups in relation to Q-values. Salmonids (1+ and older) were the best indicators of water quality in terms of species composition (%) and abundance (no. fish m^{-2}), as indicated by Q-values. Salmonids (1+ and older) divided into five distinct groups: Group 1 (Q2–3), Group 2 (Q3), Group 3 (Q3–4 and Q4), Group 4 (Q4–5) and Group 5 (Q5) in terms of species composition. In terms of abundance, the 1+ and older salmonids divided into three significantly different groups: Group 1 (Q2–3 and Q3), Group 2 (Q3–4 and Q4) and Group 3 (Q4–5 and Q5). Moreover, the abundance of 1+ and older salmon was significantly different between moderate (Q3–4) and good-quality (Q4) sites. These simple metrics 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.

Using the fish-community data generated by the project, a predictive model was developed for fish in rivers. All sites achieving a Q-value of Q4–5 and Q5 were considered 'high'-quality or possible reference sites. The observed/expected (*O/E*) scores grouped by Q-values were significantly different for the six Q-value groups; the differences were between the lower Q-values but some overlap occurred at the higher Q-values. The discriminant model assessment showed that sites were correctly assigned to bio-groups and the distribution of reference site *O/E* scores was similar to many published RIVPACS and AUSRIVAS models using fish and invertebrates. This suggests that the model produced here is robust and to the standard of other similar models in use worldwide. While the correlation with Q-scores was strong and positive, there was no significant difference between the reference sites and the Q3–4 sites.

In order to develop a robust and accurate monitoring protocol for the WFD's ecological assessment for fish in

rivers, the degree of variation in fish populations and physical habitat variables were evaluated at the reach scale. The number of sites required in a stream with low longitudinal variation is less than that for a stream with high longitudinal variation (habitat and environmental stress). Ideally, a sample length of between 300 and 450m (i.e. between 10 and 15 sample stretches of 30–45m in length) is required for a typical unpolluted gravel-bed river with a mean width of 5m. Densities of salmon and trout estimated from single-pass electric fishing correlated significantly with densities estimated by the Zippin multi-pass depletion method (Zippin, 1958). While single-pass fishing is less costly and may be adequate for the assessment of salmonids, it is considered inadequate for WFD purposes where all species must be monitored.

The project required a preliminary evaluation of fish-stock assessment in lakes using conventional gill-netting methods and hydro-acoustic techniques. Two lakes were studied, one deep and one shallow. Both a vertical and a horizontal transducer were used. The former was found to deliver satisfactory results in the deeper water but was inefficient in the shallows where significant numbers of fish were caught in the nets. Strong winds militated against the gathering of meaningful acoustic data as echo traces were obscured completely by noise from bubbles. It is therefore recommended that hydro-acoustic surveys should be conducted in early morning or at night when waters are calm and when fish may be distributed through the water column more evenly.

The effects of diurnal oxygen variation on trout and eels in rivers exhibiting different levels of enrichment were investigated using radiotelemetry. Resident fish in three rivers were tagged and their movements tracked, and dissolved oxygen (DO) in these rivers was monitored throughout the study periods. Diurnal variation was greatest in the most enriched river where oxygen dropped to critical levels (50%) at dawn on 5 consecutive days, which may have stressed the salmonids. Some individual trout remained stationary in marginal reed beds for extended periods during this time. No statistical association was made between fish movement and oxygen or temperature in any of the waters even under extreme conditions. Losses to predation were highest in the most enriched system and it is thought that stationary (possibly lethargic) trout were more vulnerable in this river at times of low oxygen.

The project also researched the variability in egg and alevin survival rates across a water-quality/nutrient gradient. Survival was highly variable and site specific. The pattern generally deviated from the expected trend of increasing mortality/decreasing survival from high to poor-quality sites. High survival rates were found at some enriched sites – however, this is no indication of whether the site could sustain a brown trout population throughout the entire life cycle.

1 Introduction

This project explores the relationship between fish communities and environmental influences that control them. The primary aim of the project was:

> … to develop a predictive model for the composition, abundance and age structure of the fish fauna based on Q-Values, faunal and floral communities, physical and hydrological environment plus environmental variables such as nutrient concentrations.

To achieve this main aim, the project team sampled fish populations at 374 river sites in Ireland, gathering extensive physical/hydromorphological, biological and chemical data (see Appendix 1). A further 145 sites were extracted from the Central Fisheries Board's (CFB) archival database. This section outlines the analysis of this dataset and the resultant relationships between the fish populations and the hydromorphological environment initially. Sections 2 and 3 take water-quality issues into account using the Environmental Protection Agency's (EPA) Quality Rating System (Q-values) and water chemistry. An integral part of a fish-assessment system is to examine the spatial and temporal variability of fish populations; Section 4 examines this aspect and recommends sampling methodologies for rivers and outlines some preliminary results for Irish lakes. Sections 5 and 6 discuss additional studies that were undertaken to answer questions raised in the project specification regarding mechanisms of water-quality impacts on fish – specifically, diurnal oxygen variation as it affects fish movements and micro-distribution and the impact of eutrophication and siltation on salmonid-egg survival.

Since 1971, the EPA has used the Quality Rating System (Q-values) to assess water quality in Irish rivers, primarily on the basis of macroinvertebrate communities in riffle areas, but also taking into account aquatic macrophytes, phytobenthos and hydromorphology (Flanagan and Toner, 1972; Clabby et al., 1992; McGarrigle et al., 2002). The Quality Rating System has been shown to be a robust and sensitive measure of riverine water quality and has been linked with both chemical status and land-use pressures in catchments (Clabby et al., 1992; McGarrigle, 1998; Donohue et al., 2006). The system facilitates rapid and effective assessment of the water quality of rivers and streams. There are nine Q-value scores, ranging from 1 to 5 (intermediate scores such as Q4–5 are also possible). High ecological quality is indicated by Q5, Q4–5 while Q1 indicates bad quality (Table 1.1). The project examined this association and developed a statistical relationship between the existing Q-rating system and fish-community structure/composition.

The Water Framework Directive (EC Directive 2000/60/EC) (WFD) lists fish amongst the biological elements which should be used for the classification of ecological status of surface waters (rivers, lakes and transitional waters [estuaries]). 'Ecological status' (Art. 2 [21]) is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V (Wallin et al., 2003). Member states are required to establish methods and tools for assessing ecological status and guidance on the approach to classification is provided (ECOSTAT, 2003).

Table 1.1: General characteristics of the various biological quality classes (after Clabby et al., 2001)

Quality classes	Class A		Class B	Class C	Class D	
Q ratings	Q5	Q4	Q3–4	Q3	Q2	Q1
Pollution status	Pristine, unpolluted	Unpolluted	Slight pollution	Moderate pollution	Heavy pollution	Gross pollution
Fishery potential	Game fisheries	Good game fisheries	Game fish at risk	Coarse fisheries	Fish usually absent	Fish absent

Twenty-nine species of fish known to occur in Irish freshwaters (Went and Kennedy, 1976; Maitland and Campbell, 1992) are listed with observations on their origin and current status (Table 1.2). Of these, the allis shad (*Alosa alosa*), twaite shad (*Alosa fallax*), smelt (*Osmerus eperlanus*) and flounder (*Platichthys flesus*) are primarily fish of estuaries and coastal waters. The sturgeon (*Acipenser sturio*) enters Irish freshwaters only very occasionally.

In Ireland, the Killarney shad (*Alosa fallax killarnensis*), char (*Salvelius alpinus*) and pollan (*Coregonus autumnalis*) are confined to lakes, as are rainbow trout (*Oncorhynchus mykiss*). Carp (*Cyprinus carpio*), tench (*Tinca tinca*), bream (*Abramis brama*) and rudd (*Scardinius erythrophthalmus*) are primarily species of standing waters but also occur in very slow-flowing deep water in some rivers (habitats not surveyed in this project).

Qualitative information on fish stocks was compiled by the Inland Fisheries Trust (IFT) from 1950 to 1979 and thereafter by the Central and Regional Fisheries Boards (CFB and RFBs). Since 1975, O'Grady's (1981) quantitative technique for assessing trout populations in managed game fisheries has been used; and trout stocks in selected rivers have also been assessed quantitatively since that time (Champ, 1983). Since the mid-1980s standard baseline habitat surveys were conducted in all catchments where instream works, drainage maintenance etc. were carried out (O'Grady et al., 1991). However, no systematic national monitoring programme exists for fish in Irish lakes or rivers.

Table 1.2: List of freshwater fish species of Ireland (scientific and common names)

Common name	Scientific name	Status	
Species which spend their entire life or the major part thereof in freshwater			
River Lamprey*	***Lampetra fluviatilis* (Linnaeus 1758)**	W	A
Brook Lamprey*	***Lampetra planeri* (Bloch 1784)**	L	C/R
Sea Lamprey*	***Petromyzon marinus* (Linnaeus 1758)**	L	C/R
Killarney Shad	***Alosa fallax killarnensis* (Regan)**	L	R
Atlantic Salmon*	***Salmo salar* (Linnaeus 1758)**	W	A
Brown Trout/Sea Trout*	***Salmo trutta* (Linnaeus 1758)**	W	A
Rainbow Trout	*Oncorhynchus mykiss* (Walbaum 1792)	L	R
Arctic Char	***Salvelinus alpinus* (Linnaeus 1758)**	L	R
Pollan	***Coregonus autumnalis* (Pallas 1776)**	L	R
Pike*	*Esox lucius* (Linnaeus 1758)	W	A
Common Carp	*Cyprinus carpio* (Linnaeus 1758)	L	C
Gudgeon*	*Gobio gobio* (Linnaeus 1758)	W	A
Tench	*Tinca tinca* (Linnaeus 1758)	L	C
Common Bream	*Abramis brama* (Linnaeus 1758)	W	A
Minnow*	*Phoxinus phoxinus* (Linnaeus 1758)	W	A
Rudd	*Scardinius erythrophthalmus* (Linnaeus 1758)	W	C
Roach*	*Rutilus rutilus* (Linnaeus 1758)	W	C
Dace	*Leuciscus leuciscus* (Linnaeus 1758)	L	R
Chub	*Leuciscus cephalus* (Linnaeus 1758)	L	R
Stoneloach*	*Barbatula barbatula* (Linnaeus 1758)	W	A
European Eel*	***Anguilla anguilla* (Linnaeus 1758)**	W	A
Three-Spined Stickleback*	***Gasterosteus aculeatus* (Linnaeus 1758)**	W	A
Ten-Spined Stickleback*	***Pungitius pungitius* (Linnaeus 1758)**	L	C
Perch*	*Perca fluviatilis* (Linneaus 1758)	W	A
Species which enter freshwater to spawn near the upstream limit of tidal influence			
Twaite Shad	***Alosa fallax* (Lacepede 1803)**	L	R
Smelt	***Osmerus eperlanus* (Linnaeus 1758)**	L	R
Species which may enter freshwater for variable periods but principally occur in marine or estuarine waters			
Allis Shad	***Alosa alosa* (Linnaeus 1758)**	L	R
Sturgeon	***Acipenser sturio* (Linnaeus 1758)**	L	R
Flounder*	***Platichthys flesus* (Linnaeus 1758)**	W	C

Note: **Native species in bold type**
* Denotes species recorded during the survey
L – Local; W – Widespread; R – Rare; C – Common; A – Abundant.

The use of fish communities as indicators for the ecological quality of running water is becoming more common worldwide (Karr, 1981; Scott and Hall, 1997; Kestemont et al., 1998; Appelberg et al., 2000; Belpaire et al., 2000; Kesminas and Virbickas, 2000; Schmutz et al., 2000; McCormick et al., 2001; Joy and Death, 2002; Mebane et al., 2003; FAME CONSORTIUM, 2004; Pont et al., 2006). However, previously, fish have been overlooked because of a number of factors: fish mobility in time and space; greater personnel needs for sampling than for other taxa; and the high costs of field sampling (Karr, 1981; Berkman and Rabeni, 1986). However, fish, which provide a dramatic impact when mortality occurs, can work as powerful tools for assessing aquatic environments and are highly suitable as indicators of human disturbances. Indeed, fish have a number of advantages as indicator organisms for biological-monitoring programmes (Karr, 1981; Harris, 1995; FAME CONSORTIUM, 2004):

- Present in most surface water, they occupy a variety of habitats and are easily identifiable.
- Most species' ecological requirements and life histories are well understood.
- Many species' sensitivity to disturbances and their response to environmental stressors are often known.
- Complex migration patterns make some fish sensitive to continual interruptions.
- The longevity of many fish species enables them to be sensitive to disturbance over relatively wide temporal and spatial ranges.
- Fish communities are valuable economic resources and the public can relate to them.

Standardised fish-based methods for assessing the ecological integrity of running waters were first developed in the USA in the 1980s (Karr, 1981; Angermeier and Karr, 1986; Karr et al., 1986; Karr et al., 1987). In Europe, fish-based methods are increasingly important now that fish are one of the four biotic elements listed in the EU WFD (CEC, 2000) on which water body status will be assessed, and a number of studies have been undertaken (Appelberg, 2000; Belpaire et al., 2000; Kesminas and Virbickas, 2000; Schmutz et al., 2000; Pont et al., 2006). A multimetric fish-based index (the European Fish Index [EFI]) was developed in 2004 by a consortium of researchers from 12 countries in Europe (Austria, Belgium, France, Germany, Greece, Lithuania, Poland, Portugal, Spain, Sweden, the Netherlands and the United Kingdom) (the FAME project) (FAME CONSORTIUM, 2004; Pont et al., 2006) based on the concept of the Index of Biotic Integrity (Karr, 1981).

The Irish freshwater fish fauna broadly consists of two distinct groups: (i) the salmonids, which require low temperatures and high oxygen and (ii) the cyprinids which prefer higher temperatures and display a range of tolerance to low oxygen. The aims and objectives of the project were:

1 To assess the impact of water quality, as evidenced by the EPA's Quality Ratings on riverine fish stocks by establishing a relationship between fish stocks in rivers and the Q-value system.

2 To develop a model, with known accuracy and precision, to predict the composition of fish in rivers (based on the integration of hydromorphological and the biotic elements of the aquatic ecosystem).

3 To provide recommendations for fish-stock assessment in rivers (and investigate methods for lakes).

4 To develop an increased understanding of the impacts of eutrophication on fish populations.

Sections 2 and 3 deal with relationships between the fish communities and the physical environment, layering water-quality aspects on top of hydromorphological factors using a range of statistical and modelling approaches. Section 4 looks at sampling variability and recommends methods for sampling of Irish river and lake populations. Sections 5 and 6 attempt to answer questions posed in the original project specification on the impact of diurnal oxygen variation (Section 5) and silt and blanket weed on spawning (Section 6). More detail on each section of the report is available in the main report (Kelly et al., 2007, www.epa.ie).

2 Establishment of a Relationship between Q-Values and Fish-Stock Composition and Abundance

2.1 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 can function as continuous monitors. Both fish and benthic macroinvertebrates have been used in water-quality assessment. Macroinvertebrates are the primary biological element currently being used in the WFD Intercalibration process (McGarrigle, pers. comm.).

The EPA Quality Rating (Q-values) System has been used to monitor the water quality of streams and rivers in Ireland since the 1970s. 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 that are most at risk of fish kills (McGarrigle, 2001).

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 the composition of fish species 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.

2.2 Methods

Electric fishing surveys were conducted at 374 river sites between 2001 and 2003 (see Appendix 1). The project also drew on archival data and on fish-stock assessment surveys being conducted independently by the fisheries boards. Sites were selected to cover the complete range of EPA Q-values (Q1–Q5). In addition, four 'core' rivers (Oona water, Rye water, Dunkellin and the Robe) where scientific investigations were ongoing or had previously been undertaken were selected for study in order to examine natural spatial and temporal variation in fish communities. These rivers exhibit different degrees of biological and physical impairment because of a variety of anthropogenic pressures. Spatial and temporal information on fish stocks at clean and impacted locations in the River Slaney catchment has also been evaluated.

2.2.1 Fish-Stock Assessment

In order to obtain a representative sample of the fish assemblage at each sampling site, electric fishing was the method used (Kelly, 2001). The technique complies with CEN guidance for fish-stock assessment in wadable rivers (CEN, 2003). Each sampling area was isolated using stop-nets and a number of fishings were carried out. Each section included all habitat types (i.e. riffle, glide, pool). All population estimates were converted to minimum densities (i.e. no. m^{-2} for combined runs) to standardise the dataset for statistical analysis.

2.2.2 Macroinvertebrates

Macroinvertebrates were assessed at each site using the standard EPA methodology. Some samples were preserved for laboratory examination and quality control. The relative abundance of each taxon identified was recorded and an EPA Q-value rating was applied to each site sampled as an indication of water quality.

2.2.3 Environmental/Abiotic Variables

A number of physical habitat variables was measured at each site to complement the species lists, for example, percentage of overhead shade, percentage of substrate type and instream cover. The percentage of riffle, glide and pool was measured over each reach surveyed. Land use, altitude, catchment area, eastings and northings, distance from source and sea, and stream order (Strahler, 1952) were calculated.

2.2.4 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 metals. Limited in situ monitoring was also carried out (i.e. dissolved oxygen, temperature and conductivity).

2.2.5 Quality Assurance/Q-Value Validation

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

2.2.6 Data Analysis

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

Two-way indicator species analysis (TWINSPAN – Hill, 1994), a hierarchical classification program, was used to classify sites according to their species composition. Validation of the fish and macroinvertebrate end groups in TWINSPAN was verified using multi-response permutation procedure (MRPP) in PC-ORD (McCune and Mefford, 1997) using Euclidean distance measures.

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). Both groups were then compared using a Mantel test as a final test of concordance (Clarke and Gorley, 2001).

Detrended correspondence analysis (DCA – ter Braak and Prentice, 1988) was used as an initial indirect ordination technique. 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 all canonical axes was tested using Monte Carlo permutation tests (with 9 999 permutations). The taxa scores and the correlations between environmental variables and the axes were plotted on two-dimensional graphs, i.e. biplots.

2.3 Results

A total of 519 sites is included in the database: however, only 470 of these have been subjected to univariate analysis and 394 to multivariate analysis owing to incomplete datasets.

2.3.1 Q-Values Versus Altitude

Q-values were located over a range of altitudes, ranging from 5.5m (Q2–3) to 263.8m (Q4). A number of the high-quality sites surveyed during the project were located in lowland areas (high quality, minimum altitude = 8.2m). Statistical analysis showed that there was a significant relationship between altitude and Q-value (one-way ANOVA, df = 471, F = 4.59, P = 0.001). 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.

2.3.2 Spatial Variation on Core Rivers

All sites on the Oona water 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 16 and 17 May 2002 when dissolved oxygen (DO) remained at or below 5.0 mg l^{-1} for 9 to 10 hours (minimum 3.68 mg l^{-1}) but no evidence of external pollution was detected (Jordan, pers. comm.).

Brown trout and salmon stocks on the Rye water were surveyed in 2000 and 2003. Brown trout fry density ranged from 0.001 to 0.12 m^{-2}. Salmon fry density ranged from 0.001 to 0.085 m^{-2}. Brown trout density (1+ and older) varied from 0.05 to 0.31 m^{-2} and remained higher than salmon parr densities (0.001 to 0.12 m^{-2}) in both stretches. Water quality remained at Q3 throughout.

A slight deterioration in quality was noted from 2001 to 2002 on the Dunkellin river. Ten fish species were recorded in 2001 and 2002; perch, present in 1986, were not encountered. Species richness of the site fish assemblages was positively correlated with stream order in all three surveys (rs = 0.81; 0.76 and 0.68 respectively for 1986, 2001 and 2002 surveys with p<0.01).

Trout were the most commonly observed species followed by three-spined stickleback, minnow and stoneloach on the Robe river. Water quality and salmonid densities fluctuated significantly at most sites over the three years of the survey. Fish were absent or three-spined stickleback dominated sites when water quality was poor, trout colonised most locations when water quality improved.

Fish stocks at each site (three clean and two impaired) in the Slaney catchment were surveyed five times between 1992 and 2003. The average densities of salmonids at the three clean (Q4, 4–5 & 5) survey sites were 0.559, 0.590 and 0.587 m^{-2}. The average densities of total salmonids were 0.610 and 0.641 m^{-2} for the two impaired sites (Q3, 3–4).

2.3.3 Relationship between Fish and Q-Values

Fish Species Richness in Relation to Q-values and Altitude

Overall, a total of 16 species of fish were recorded at the 470 sites included in the dataset. Brown trout were the most common fish species, followed by salmon, eel, three-spined stickleback, stoneloach and minnow. Juvenile lampreys, although widely distributed, were recorded at only 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.

Fish species richness was calculated for each Q-value group. The highest fish species richness was recorded at Q3–4 sites (mean = 3 species) (Fig. 2.1).

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. 2.1). The maximum number of species recorded at any one site was 8 (Q3 and Q3–4 sites). There was a significant difference in species richness between Q-values (one-way ANOVA, df = 510, F = 4.854, P = 0.001). 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 vs Q2–3 p = 0.01, Q2–3 vs Q3 p = 0.0001, Q2 vs Q3–4 P = 0.0001, Q2 vs Q4 P = 0.0001, Q2 vs Q4 p = 0.0001 and Q2 vs Q5). The mean number of species present at Q2–3 sites was also

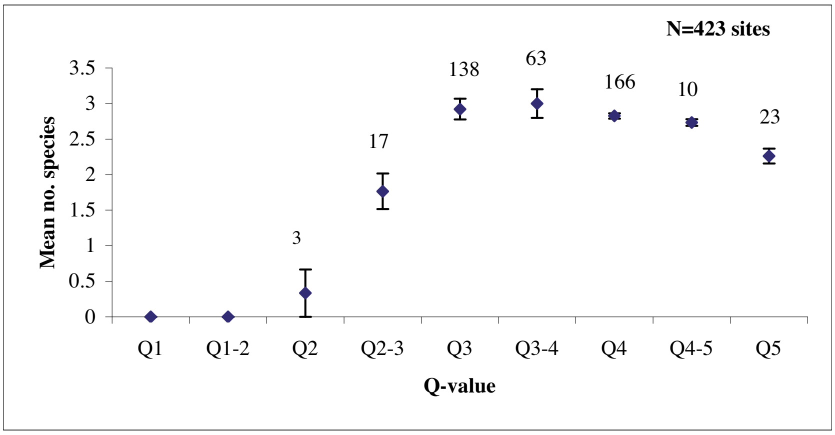

Figure 2.1: Fish species richness (including standard error) in relation to Q-values (Numbers of sampling sites in each Q-value group are shown)

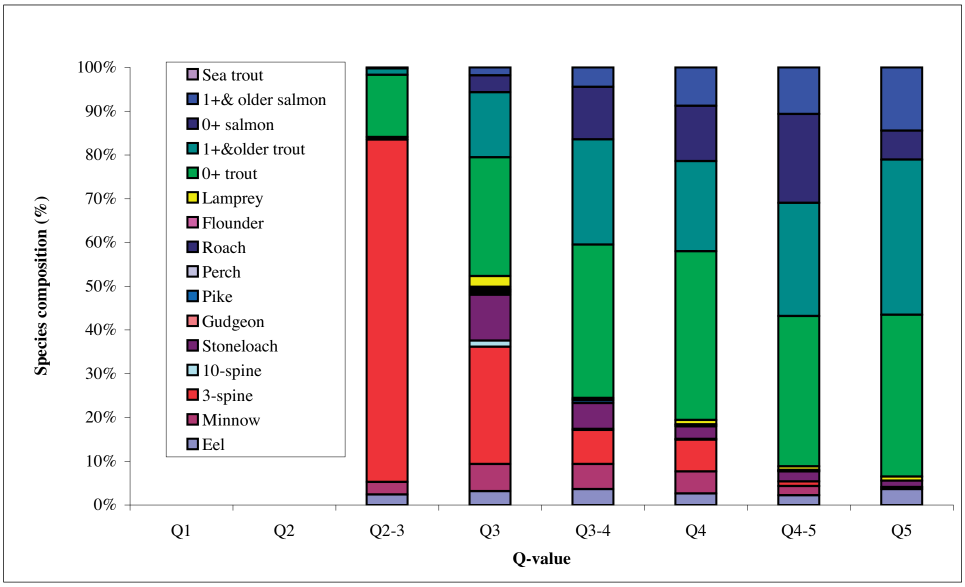

Figure 2.2: Percentage composition of fish species in relation to Q-values

significantly lower than at four other Q-value site groupings (LSD post-hoc test, Q2–3 vs Q3 p = 0.002, Q2–3 vs Q3–4 p = 0.003, Q2–3 vs Q4 p = 0.001 and Q2–3 vs Q4–5 p = 0.006 but not at Q5 sites (P = 0.13).

In general, fish-species richness decreased with increasing altitude. 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 + and older trout). Most species occurred at altitudes less than 100m. Eel, minnow, stoneloach, three-spined stickleback, lamprey, brown trout and salmon were common at all sites less than 250m. Coarse fish such as pike, perch, roach and other fish species such as ten-spined stickleback and gudgeon were absent or rare at altitudes over 150m.

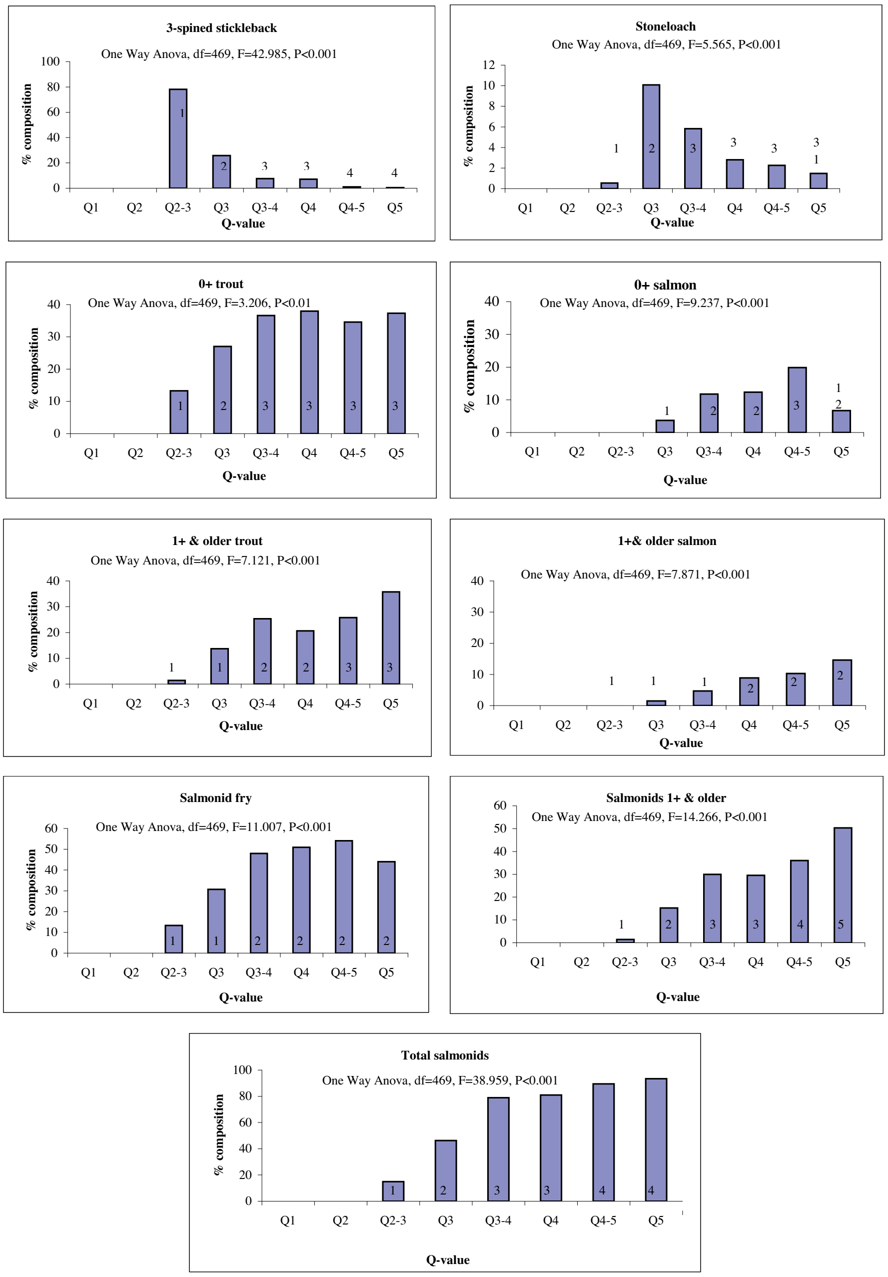

Figure 2.3: Percentage composition of selected fish groups in relation to water quality as indicated by Q-values

2.3.4 Fish Species Composition in Relation to Q-values

Fish-species composition was calculated for each site for each Q-value. Brown trout occurred at 90% of the sites, followed by salmon (41.3%), eel (39.41%) and three-spined stickleback (38.6%). Flounder were the least common (0.42%) (Fig. 2.2).

Results indicate that salmonids (trout and salmon were treated separately and in combination to allow for the absence of salmon upstream of impassable barriers) were the dominant species at Q3 to Q5 sites whereas three-spined stickleback were the dominant fish species at the more polluted sites (i.e. Q1 to Q2–3). Statistical analysis (one-way ANOVA) showed that water quality, as indicated by Q-values, had a significant effect on the percentage composition (log [x+1] transformed) of the fish community, particularly three-spined stickleback, stoneloach and all salmonid groups (Fig. 2.3).

Analysis showed that the percentage composition of three-spined stickleback was highest at Q2–3 and decreased as water quality improved (Fig. 2.3). LSD post-hoc tests divided three-spined stickleback into four distinct groups: Group 1 (Q2-3), Group 2 (Q3), Group 3 (Q3-4 and Q4) and group 4 (Q4-5 and Q5). 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. 2.3). 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. 2.3). 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. 2.3). A graph was produced illustrating the relationship between two indicator groups (non-salmonids and total salmonids) over a water quality gradient as indicated by Q-values (Fig. 2.4). This shows that non-salmonid fish species dominate the fish community at Q2–3 sites and gradually decrease to less than 10% of the fish population at Q4–5 and Q5 sites.

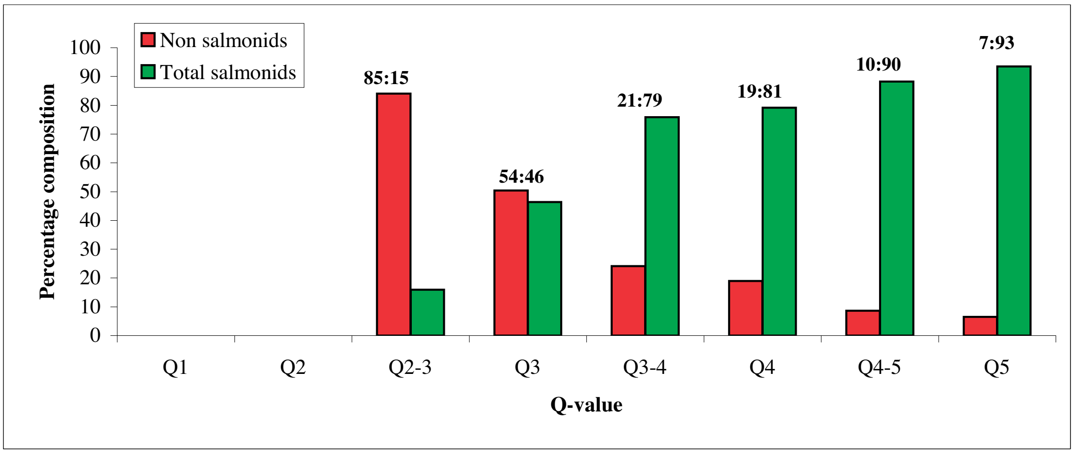

Figure 2.4: The relationship (including ratios) between salmonids and non salmonid fish species over a water quality gradient, as indicated by Q-values

2.3.5 Fish Species Abundance in Relation to Q-Values

Fish species abundance (no. m^{-2}) was calculated for each site at each Q-value (Fig. 2.5). Salmon and trout were treated separately and also combined as total salmonids for the reasons stated earlier. Two additional fish groups were also used, i.e. salmonid fry and salmonids 1+ and older. One-way ANOVA showed that water quality had a significant effect on the abundance of the fish community, particularly, three-spined stickleback and the seven salmonid groups. Three-spined stickleback decreased and salmonids, in general, increased with increasing water quality. LSD post-hoc tests indicated that the abundance of three-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. 2.5). Statistical analysis also showed that abundance (no. m^{-2}) of the salmonids (1+ and older group) was also a good indicator of water quality. Abundance of salmonids (1+ and older) was significantly higher at Q4–5 and Q5 than at any other sites (Fig. 2.5). LSD post-hoc tests divided this taxonomic group into three significantly different groups: Group 1 (Q2–3 and Q3), Group 2 (Q3–4 and Q4) and Group 3 (Q4–5 and Q5). LSD post-hoc tests also showed that abundance of 1+ and older salmon was significantly different between sites rated moderate (Q3–4) and good (Q4) quality.

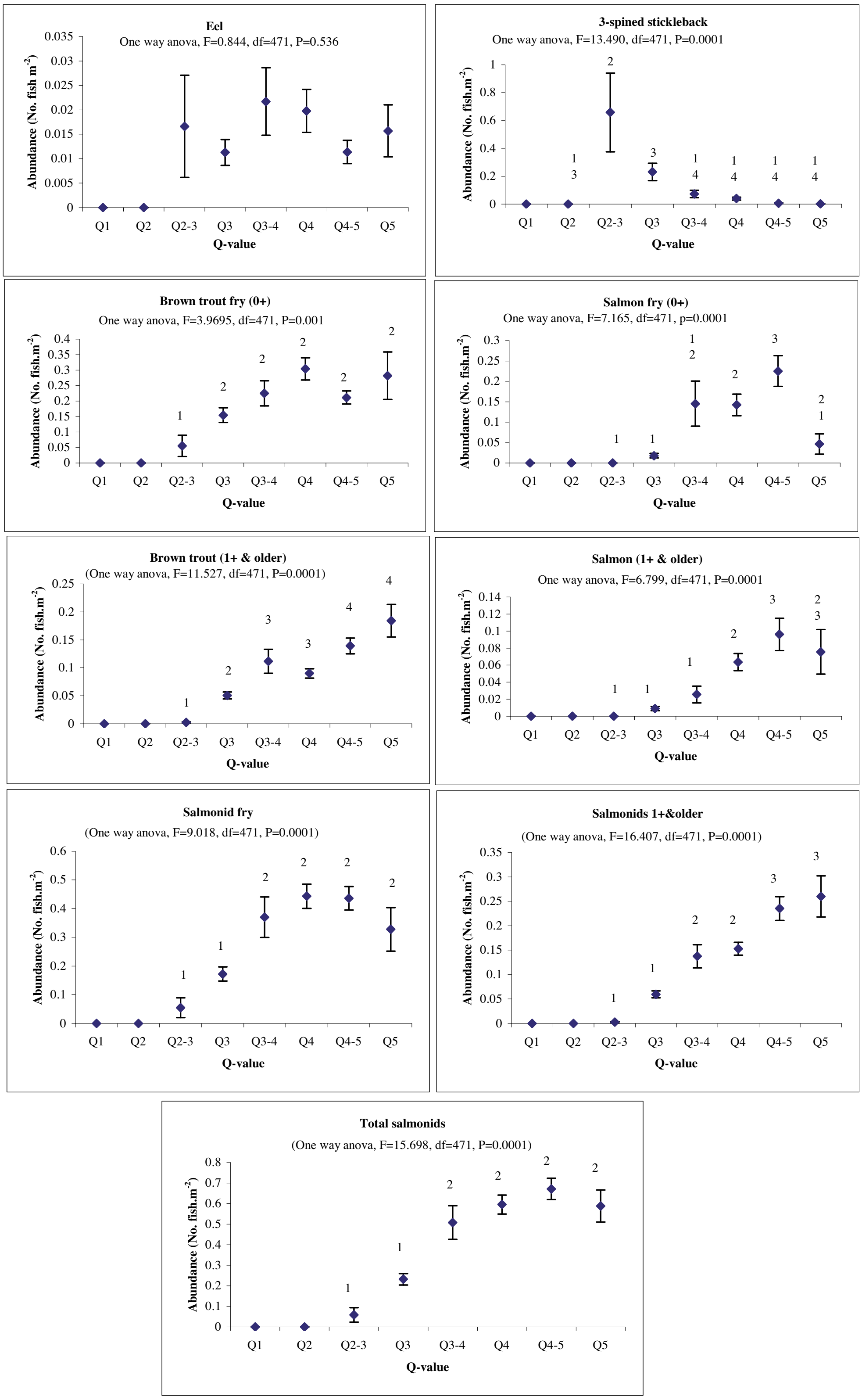

Figure 2.5: Abundance (No. fish.m-2) (indluding standard error) of selected fish groups in relation to EPA Q-values. Groups with the same number are not significantly difference at the 5% level (One-way ANOVA and LSD post-hoc test)

2.3.6 Classification of Fish Using TWINSPAN

TWINSPAN was carried out on 393 sites and produced 36 site groupings (using quantitative data). A TWINSPAN division was only accepted if the groups differed significantly ($P<0.05$) according to MRPP (PC-ORD), and this resulted in 14 final fish groups (Fig. 2.6). The first level of the TWINSPAN hierarchy separated the sites into two groups of 357 and 28 sites. The 28 sites to the right of the TWINSPAN tree differed owing to the presence of high densities of three-spined stickleback. The second level divided the 357 sites into 172 sites and 185 sites. The 185-site grouping differed because of 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 of the TWINSPAN tree further subdivided by the presence of salmon (Fig. 2.6).

2.3.7 Classification of Macroinvertebrates Using TWINSPAN

TWINSPAN was then carried out on 391 sites (using abundance data) and produced 237 site groupings. After MRPP analysis, 37 final macroinvertebrate groupings remained. In general, the grouping of the macroinvertebrate sites reflected the Q-value gradient as the majority of clean sites (Q4–5 and Q5) pulled to the left of the dendrogram and most sites that were enriched pulled away to the right of the dendrogram. However, the separation was not as clear for sites rated Q3 to Q3–4.

2.3.8 Concordance of Fish and Macroinvertebrates

The 37 macroinvertebrate groupings were imposed on the fish abundance dataset and the 14 fish groupings were imposed on the macroinvertebrate data using ANOSIM to determine whether the classification for one group would apply to the other. Macroinvertebrate TWINSPAN end groupings were statistically significant ($R = 0.633$, $p = 0.001$) (Table 2.1). There was also a statistically significant separation when these groupings were imposed on the fish data, although the R value was low ($R = 0.238$, $P = 0.001$) (Table 2.1). 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 2.1), indicating that other factors are structuring the fish communities and that the impact of water quality acts on top of these.

2.3.9 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 indicates that there is a positive association between the fish and macroinvertebrate matrices.

2.3.10 Environmental Definition of Macroinvertebrate and Fish Sites using Ordination

The pattern of variation in fish and macroinvertebrate community composition at 390 sites in relation to 28 environmental parameters (physical and chemical) was analysed using CCA. Both biotic groups were subjected to an identical environmental matrix.

Table 2.1: 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

*ns =not significant

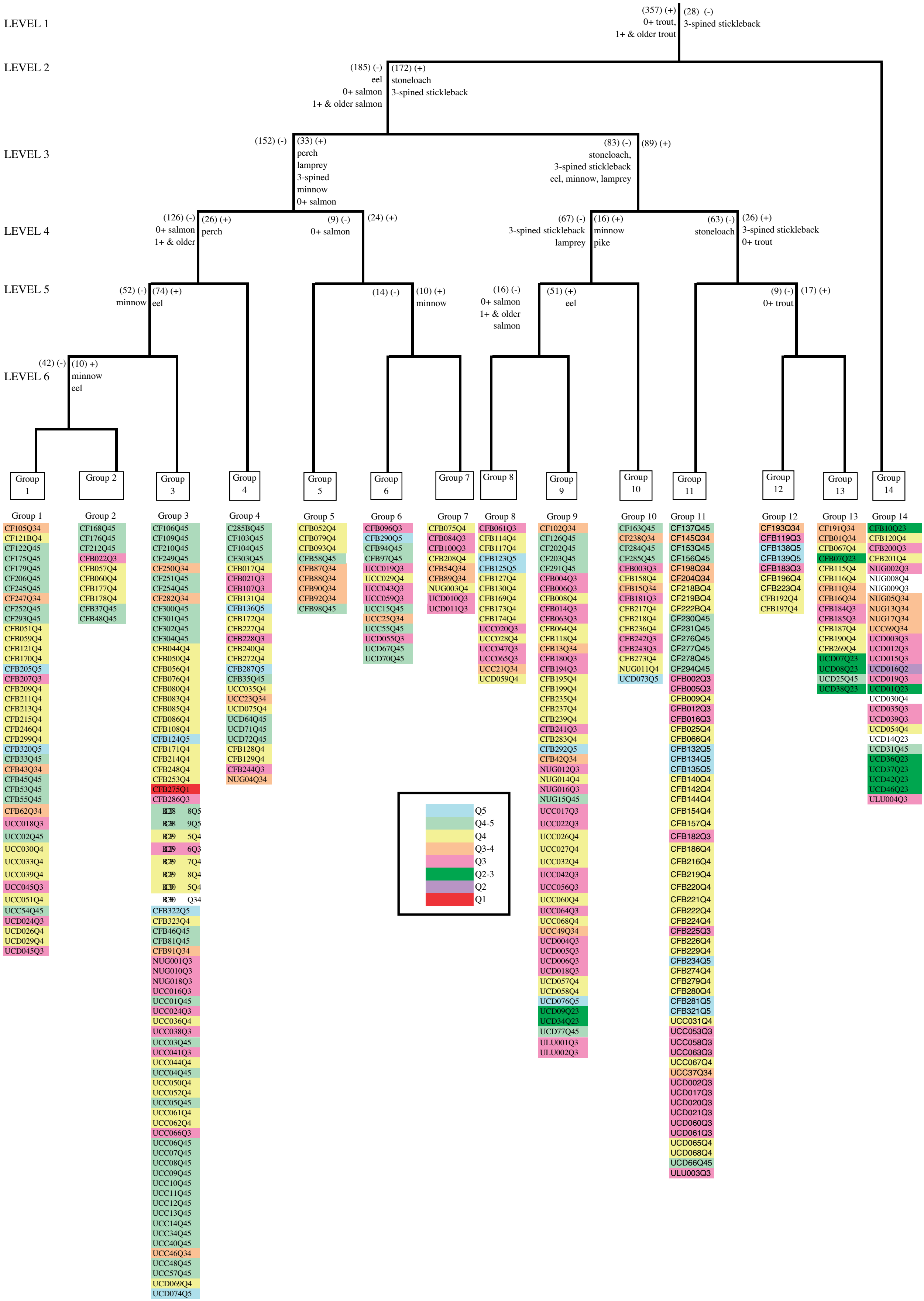

Figure 2.6: TWINSPAN classification of fish species at 385 sites surveyed during the project

Analysis of the fish data in relation to the environmental variables indicated that alkalinity, barrier downstream and dimensions of the site (represented by surface area) were the three most important variables influencing the fish community composition. Similarly, alkalinity, northing, the presence of a barrier to fish migration downstream of the site and geology were found to be the most important variables influencing the macroinvertebrate community. The scores for species and environment (0.7395 and 0.7976 respectively) indicate that most of the variation in fish and macroinvertebrate compositions is caused by the environmental variables. The first canonical axis for fish accounts for 14% of the variation; however, the first axis accounts for only 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).

Fish species, such as three-spined stickleback, indicative of enriched sites, are located on the right-hand side of the biplot. 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 correlated negatively with barrier downstream of the site. In general, many of the more enriched and organically polluted 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. 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.

Figure 2.8 shows 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 as indicated by Q-values were 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 are associated with mean wetted width, percentage boulder, stream order, percentage riffle and altitude. Intolerant macroinvertebrate taxa were situated on the right-hand side of the biplot and tolerant taxa on the left-hand side of the biplot.

2.4 Discussion

The present study surveyed fish populations at over 500 locations in 1st to 6th order streams (mostly in small to medium wadable rivers, only 10 sites were stream order 5 and 6) across the full range of Q-values. Native species (apart from flounder) were well distributed at all elevations (0–263m), trout were the most widespread, occurring at 83%–100% of sites. Non-native species exhibit a sporadic distribution in Irish rivers and are absent from many catchments; pike and perch were recorded at 12% of sites, gudgeon at 5% of sites and roach at 1% of sites. The study showed that high-quality sites (Q4–5 and Q5) were more common at higher altitudes than poor-quality sites; however, a number of high-quality sites were located in lowland areas. In general, in many European countries, 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 distinct) biological zones, each of which has a characteristic fish fauna with a diagnostic or 'key' species, i.e. (i) trout, (ii) grayling (*Thymallus thymallus*), (iii) barbel (*Barbus barbus*) and (iv) bream. Huet (1959) further states that the four zones really represent two faunistic regions, i.e. an upper salmonid region of cooler waters and a lower cyprinid region of warmer waters. The fish zonation is mostly the result of the physical characteristics of stream gradient, stream width, current speed and temperature. Irish rivers are mostly short and fast flowing; temperature rarely exceeds 20°C and salmonids normally dominate stocks from headwaters to the sea. Bream have been introduced but are not widespread and, with the exception of the River Shannon, they are confined to the very lower reaches of the few rivers in which they occur; grayling and barbel do not occur in Ireland. Consequently, this change in fish community suggested by Huet (1959) does not exist in the vast majority of Irish rivers and cannot occur in the many catchments where cyprinids have not been introduced. In Ireland, the geographical distribution of coarse fish species is patchy but is expanding – roach, dace, chub (a recent introduction) and bream may eventually populate more systems.

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; Cowx and Welcomme, 1998). This is also indicated by the correlation of fish in Section 3 with distance from source, wetted area, stream order, etc. Ordination analyses 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. The significance of physical factors on fish distribution and abundance is also described in Section 4. Nonetheless, biota (including fish) will change because of external pressures independent of hydromorphological factors. This is apparent from the occurrence of fish kills following severe pollution events and the lack of fish in chronically polluted sites that are otherwise hydromorphologically suitable (Champ, 2000).

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 Q-values (Q1– Q5) and to assess the feasibility of using fish assemblages as biological indicators of river water quality in Irish rivers. The EPA Q-value system is 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). Whilst some impairment is evident at Q4, the ecological conditions at such locations are considered to be acceptable to salmonids. 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 that it was statistically possible to separate a number of fish groups in relation to a number of Q-value groups. Two salmonid groups, i.e. total salmonids and salmonids (1+ & older), were identified as the best indicators of water quality, in terms of species composition (%). Total salmonids divided into four statistically different groups: Group 1 (Q2–3), Group 2 (Q3), Group 3 (Q3–4 and Q4), and Group 4 (Q4–5 and Q5) (Fig. 2.3). Salmonids (1+ and older) divided into five distinct groups: 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 this taxonomic group into three significantly different groups: Group 1 (Q2–3 and Q3), Group 2 (Q3–4 and Q4) and Group 3 (Q4–5 and Q5). Statistical analysis also showed that abundance of 1+ and older salmon were significantly different between sites rated moderate (Q3–4) and good (Q4) status.

In this study, there was a continuous decrease in salmonid abundance in relation to decreasing ecological quality, with loss of salmonids, at or below Q3. 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 which may have colonised from cleaner, neighbouring habitats. Trout and salmon occurred at enriched poor-quality (Q3) sites through to unpolluted high-quality sites (Q5). Salmonids, though present at poor-quality sites, are at risk and may be unsustainable in the longer term because of occasional pressures associated with the additive effect of high temperature and oxygen, as prolific growths of benthic macroalgae are characteristic features at such quality ratings (McGarrigle, 2001). The most significant fish community change occurred between poor-quality Q2–3 and Q3 sites, particularly in relation to the reduction in salmonid representation at the lower Q-values and the increase in three-spined stickleback. The three-spined stickleback appears to be relatively pollution tolerant (McCarthy and Kennedy, 1965; Maitland and Campbell, 1992) and is also a good coloniser of rivers recovering from severe pollution (Turnpenny and Williams, 1980).

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 decrease slightly to high-quality (Q5) sites (mean = 2 species). The maximum number of species recorded at any one site was 8 (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, three-spined stickleback, lamprey, stoneloach, minnow, roach and eels. These results concur with the findings of Vannote et al. (1980) and Miltner and Rankin (1998), i.e. the abundance of fish and macroinvertebrates is generally highest at intermediate nutrient levels. Miltner and Rankin (1998) also found that the relative abundance of tolerant and omnivorous fish increased significantly in relation to nutrient enrichment in headwaters, wadable streams and small rivers. A low species richness was recorded at high-quality Q4–5 and Q5 sites in this study which McDonnell (2005) linked to stream order and altitude, stating that 'species richness' may not be a good indicator of water quality in Irish streams owing to the paucity of fish species present. Angermeier and Davideanu (2004) created an 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 DO and siltation of spawning gravel (Lee and Jones, 1991). Where cyprinids have not been introduced or colonised naturally, as is the case in many Irish river channels, clear distinctions are likely to be less evident. There is no linear relationship between species richness and water quality in rivers in Ireland and while this is unsurprising, there is nevertheless a relationship. 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, whilst 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, being particularly at risk in warm summers due to severe fluctuation in oxygen at such locations (McGarrigle, 2001).

The fish densities and species richness on the Oona water were so low as to be of little value in the context of the spatial and temporal variation study. There was no clear relationship between salmonid densities and water quality on the Rye water, which fluctuated between Q3 and Q3–4 over the ten-year period for which data were available. The results of the Dunkellin surveys and earlier data provide a baseline against which to evaluate future changes. 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 linked loosely 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 enter sites when water-quality improvement occurred. Biological assessment of water quality was not conducted at the same locations as the fish stock survey on the Slaney (only at two of the five sites) nor were these surveys conducted concurrently. However, abundance of total salmonids appears to be more stable at the better sites (Q4–5 and Q5) on the Derreen, Upper Slaney and Clody rivers. In contrast, the Bann and Boro rivers, where quality is mostly less than good status (Q3–4 to Q4), fish abundances oscillated more widely and this is attributable mainly to pulses in salmon fry abundance. It does appear that there is not a strong relationship between densities of total salmonids and Q-values in the Slaney dataset. Elliott (1994) warns of the inadequacies of short-term studies for evaluating 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 wide fluctuations in density were noted at the Bann and Boro sites that may not be related to water quality but to spawning effort. These two locations, with the highest stock density and the widest temporal range, are situated on the tributaries where water quality is impaired (Q3, Q3–4 and Q4). The mean abundance for total salmonids at all five sites over the ten-year study period was 0.6 salmonids m^{-2}.

The results from the present study show that there is significant concordance between fish and macroinvertebrate community composition in streams across Ireland. Kilgour and Barton (1999) found a significant association between fish and macroinvertebrates in Canadian streams, as both communities were responding to similar environmental variables. In the current study, both biotic groups were responding largely to similar environmental gradients as shown by CCA; these included alkalinity, barrier downstream and northing. Concordance between the two biotic groups suggests that it may be feasible to use macroinvertebrate community structure to assess fish community across a water pollution gradient. Although there is concordance between the two groups there is, however, considerable overlap in the fish species abundance (and densities of salmonids) found at all sites rated Q3 and above.

Fish community structure was related to (i) alkalinity, (ii) the occurrence of barriers downstream of the survey site and (iii) wetted width. Pollution-tolerant fish species such as the three-spined stickleback were grouped to the right of the biplot, and were associated with alkalinity and mud and silt. Salmon were grouped to the left of the biplot and were associated with mean wetted width and stream order. Juvenile salmon were correlated negatively with barrier downstream of the site. The productivity of fish populations has been linked to the productivity of benthic communities in rivers (Krueger and Waters, 1983) and 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 of the site, northing, geology, glide and percentage instream cover were the main habitat variables, characterising the macroinvertebrate community composition. The effects of nutrient enrichment on macroinvertebrates are well documented (Mason, 1996; Parr and Mason, 2003). McDonnell (2005) found that diversity and evenness of macroinvertebrates were reduced at eutrophic sites and found that width, altitude, instream vegetation, velocity and percentage silt were the main variables influencing community composition.

According to the CCA analysis there were a number of physical habitat variables, that affected each biotic group uniquely. 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 and the large degree of divergence between the biotic groups.

Results indicate that the fish abundance metric (i.e. number of fish m^{-2}) does not separate sites according to water quality as effectively as the alternate metric, i.e. fish species composition. The findings also indicate that abundance of salmon (1+ and older) could be used as an indicator to separate ‘moderate’ and ‘good’ quality sites downstream of impassable barriers.

There is no linear relationship between 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) on aquatic habitats. Karr (1981) pioneered the use of fish communities in an Index of Bioticl 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; 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. 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.

2.5 Conclusions

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 10% 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 dataset for wadable rivers (mostly 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.

Whilst the original objectives of the project do not specifically include the WFD, the project nonetheless provides certain outputs which are useful for the directive, e.g. classification systems for fish in Irish rivers and a comprehensive geographical information system (GIS) database for fish and other priority elements. 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 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 (useful only for sites downstream of impassable barriers).

2.6 Recommendations

Further work should include the:

- 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 identify problem areas.

A potential fifth metric has also been identified, where fish composition (a simple metric of species richness weighted for species tolerances) could show a clear relationship between Q-value groupings. This also requires further investigation.

3 Development of a Predictive Model for Fish in Rivers

3.1 Introduction

Three major approaches to the biological assessment of the ecological effects of pollution and landscape alteration on streams have been developed since the 1980s (Joy and Death, 2002). One approach is multimetric, where a number of individual indices are combined to measure biotic condition e.g. the IBI (Karr, 1981; Gerritson, 1995), the ICI (Plafkin et al., 1989 for invertebrates-USEPA rapid bioassessment protocols) and the European Fish Index (EFI) (FAME CONSORTIUM, 2004). A second approach is predictive and compares fauna to those predicted by empirical models to occur in the absence of human impacts, e.g. RIVPACS (Wright et al., 1984; Clarke et al., 1996; Norris 1996), AUSRIVAS (Parsons and Norris, 1996), HABSCORE (Milner et al., 1995). For instance, a regional predictive model of freshwater fish occurrence using 200 reference sites has been developed in New Zealand (Joy and Death, 2002). This second approach is the approach used in this study. The third approach uses artificial intelligence (AI) techniques. This relatively new approach to river quality monitoring is based on attempts to model the processes of human experts using two complementary techniques: (i) pattern recognition and (ii) plausible reasoning (Walley and Fontama, 2000).

3.1.1 Predictive Modelling Process

The predictive modelling approach used here involves classifying the reference sites into groups based on the fish communities and then building a discriminant model to classify sites into these groups using the site environmental variables. This model is then used to associate test sites with suitable reference sites so that a comparison can be made between observed and expected communities. Ideally, the variables used to associate the test sites with reference sites should not be influenced by human impact so that the predictions are the fish communities to be expected in the absence of impacts. However, the predictions are based on existing conditions so the expected communities are realistic. Once the scores are found for the reference sites, they then give the basis for background or existing conditions and these become the criteria by which test sites can be assessed. The rest of the sites can then be run through the model and Observed (O)/Expected (E) scores calculated to enable comparison with biotic scores.

The Reference Condition Approach

The reference condition approach is the basis of the RIVPACS and AUSRIVAS models. It is based on comparing a biological community found at a test site to the range of communities observed at a set of reference sites. It involves the selection of a large number of sites to represent the acceptable condition of the region.

The aim of this work package was to develop a predictive model, with known accuracy and precision, to predict the composition of fish in rivers (based on the physical and biotic elements of the aquatic ecosystem).

3.2 Methods

The statistical procedures/methods used in constructing RIVPACS and AUSRIVAS type predictive models using macroinvertebrates and fish have been described elsewhere (Wright et al., 1984; Moss et al., 1987; Wright et al., 1993; Clarke et al., 1996; Smith et al., 1999; Simpson and Norris, 2000; Joy and Death, 2002) and the implementation described below followed similar steps using elements from both procedures.

3.2.1 Study Area

Reference sites were selected using EPA quality ratings (Q-values). All sites achieving a Q-value of Q4–5 and Q5 were considered ‘high’-quality or possible reference sites. The Q-values were validated by EPA biologists in conjunction with CFB staff.

3.2.2 Sampling Regime

The standard methodology for the project includes fish-stock assessment using electric fishing, kick sampling for macroinvertebrates, hydrochemical analysis and physical/habitat survey methodologies and has been described in the previous section. Surveys were carried out between July and September (to include 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). A total of 118 'high'-quality sites was used (Q4–5 and Q5). Fish-abundance data (no. fish m^{-2}) for 118 sites was used to develop the model 'with barriers' (i.e. this combines all 'high'-quality sites irrespective of whether or not a barrier or impediment to free passage existed downstream). Seventy-eight sites were used to develop the model 'without barriers' (i.e. only 'high' quality sites where no impassible barriers to fish migration were located downstream were included). The barriers were identified using a GIS based data model for the quantification of the freshwater salmon habitat asset and for the determination of the quantity of habitat available to migratory salmonids (McGinnity et al., 2003).

3.2.3 Statistical Analysis-Predictive Modelling

The 'high'-quality sites (Q5 and Q4–5) were divided into groups with similar fish communities using two-way indicator species analysis (TWINSPAN) (Hill, 1979) using PC-ORD (McCune and Mefford, 1997). This process classifies both samples and species simultaneously and is based on hierarchical divisions of reciprocal averaging ordination space. The TWINSPAN analysis was taken to two levels and then two groups were combined at the second level as group sizes were too small and this resulted in three groups.

Discriminant analysis (SAS, 2000) was used to determine how well environmental variables account for the structure of biological groupings. Variables were entered as both 'log transformed' and 'not transformed' and the transformed variables were used if they improved the crossvalidated classification rate (Clarke et al., 1996). Crossvalidation was used to check whether sites were allocated to their correct groups. The crossvalidation process (also known as 'jack-knife' or 'leave-one-out' validation) involves leaving out each site in turn, then rebuilding the model, and testing the held-out site to assess whether the site was predicted as belonging to the correct group. A site was considered to be classified correctly if the probability of belonging to the correct group is higher than it is for the other groups.

After optimisation of the discriminant functions, which allocate sites to the predetermined biological classification using their environmental characteristics (the classification is based on the assessment of the optimised suite of environmental characteristics), the next step was to predict the fish communities expected at a test site. To predict the assemblage expected at a site, the frequency with which individual taxa occur in each TWINSPAN group (i.e. the relative group frequency) was calculated as the number of sites where that taxon occurs divided by the total number of sites in the group. This is referred to as the probability of finding that taxon in that group. The overall probability of finding a taxon at a site is the relative group frequency weighted by the probability of membership in each of the three groups (Table 3.1).

Table 3.1: Example of RIVPACS type prediction of assemblage at a site

TWINSPAN group	Prob. site group membership	Prevalence in group	Prob. of species occurrence in group
A	0.7	0.8	0.56
B	0.2	0.5	0.10
C	0.1	0.2	0.02
Combined probability that species will occur at site			**0.68**

In this example, the species would be predicted to be present in the site based on a decision threshold of 0.5

The final step in site assessment is to compare observed and expected faunas. The predicted fauna was compared with the observed taxa list following the procedure originally described by Wright et al. (1984). The probabilities of the predicted taxa were summed to give the ‘expected number of taxa’ (E). The number of species actually captured at a site, providing they were predicted to occur, is the ‘observed number of taxa’ (O). The ratio of the observed to the expected number of taxa (*O/E*) and taxonomic composition is the output from the model (Moss et al., 1987).

The number of taxa observed at ‘high’-quality sites were compared with model predictions generating a distribution of reference site *O/E* ratios. Low *O/E* ratios are used to indicate sites under stress, while high ratios indicate sites with more species than expected, which may indicate sites of high conservation value (Wright, 1995). Determination of whether a site is impacted is judged based on the site’s *O/E* ratio compared with the distribution of *O/E* ratios for the reference sites.

To test for differences between the environmental site descriptors of the three TWINSPAN groups, MRPP (McCune and Mefford, 1997) using Euclidean distance measures was used.

To validate the output from the model, *O/E* ratios were calculated for sites not used in the model construction and these values were then compared with Q-scores for the sites. The differences in *O/E* scores in the Q-score groups were examined using the Kruskal-Wallis test (NPAR1WAY procedure of [SAS, 2000]) and medians were compared using Tukey’s multiple range tests with the significance level set at 5%.

3.3 Results: Model Development of Reference Sites With and Without Barriers to Fish Migration Present Downstream

3.3.1 Fish Assemblages

Removal of rare species resulted in 11 taxonomic units used for model construction (Table 3.2). From this dataset, 3 main groups were identified using TWINSPAN analysis (after validation using MRPP): Group 1 sites contained no salmon but trout occurred at all 47 sites. There were 56 sites in Group 2 and trout and salmon occurred at all of the sites; however, stoneloach was absent from all sites and three-spined sticklebacks were rare occurring at only 5% of the sites. Group 3 contained only 11 sites, and stoneloach, trout and salmon occurred at all sites in this group. Three-spined sticklebacks and lamprey were more common in this group than the other two groups.

Table 3.2: Percentage of each taxon in each of the three groups

Species	Group 1	Group 2	Group 3
Eel	30	59	55
Minnow	19	9	9
Three-spined Stickleback	11	5	64
Stoneloach	26		100
Lamprey	11	4	45
Trout	100	100	100
0+ Trout	100	91	64
Older Trout	94	96	82
Salmon		100	100
0+ Salmon		89	64
Older Salmon		89	100

3.3.2 Relationships between Fish Assemblages and Physical/Chemical Data

The TWINSPAN groups were then applied to the environmental variables and discriminant functions used to assess if the biological groupings were separable by the site descriptors. Using discriminant analysis, 87 (74%) of the 114 reference sites were assigned correctly to their predetermined biological groups using crossvalidation (Table 3.3). The highest error rate for the crossvalidated classification rate was for the smallest group (3) with only 5 (46%) of the sites classified correctly.

Table 3.3: Crossvalidated number and percentage of sites correctly classified into each of the three TWINSPAN groups by linear discriminant analysis using the concurrently measured environmental variables listed in Table 3.4

	Predicted group membership (to group)			
Group (from group)	1	2	3	% of sites correctly predicted
1	36	8	3	76
2	5	46	5	82
3	0	6	5	46

The environmental variables associated with the sites from the TWINSPAN analyses are summarised in Table 3.4. The MRPP analysis revealed significant differences between these groups based on the environmental site descriptors (T = -6.91, P <0.0001).

Correlations between the discriminant factors and the environmental variables revealed the relationships between the environmental variables and the three groups (Table 3.4 and Fig.3.1). Canonical axis 1 which accounted for 74% of the variation was strongly influenced by northing, stream order, sand, stream width, and geology. Canonical axis 2, which accounted for a further 26% of the variation, was correlated with distance to tidal limit, the proportions of mud and silt, glide, total conductivity, hardness and alkalinity.

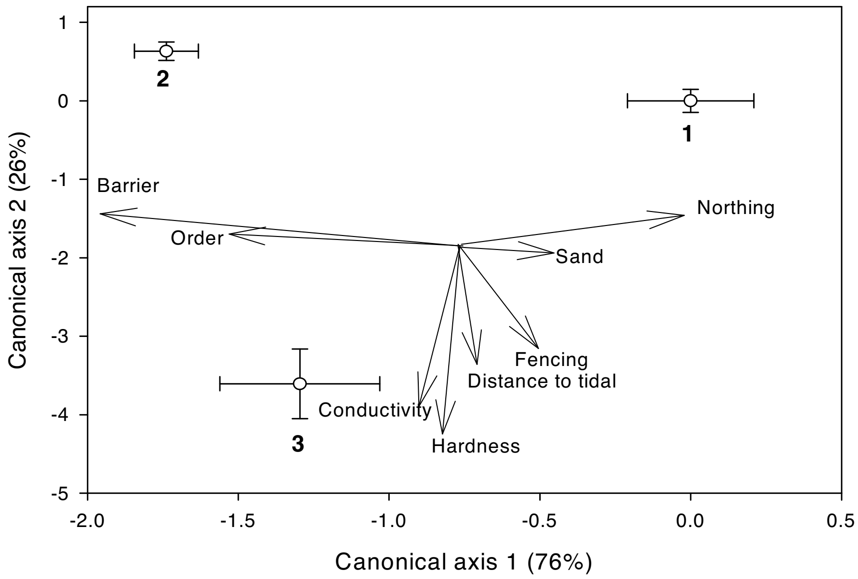

Figure 3.1: Position of the mean co-ordinates of the three TWINSPAN groups in discriminant space obtained by linear discriminant analysis on the environmental variables in Table 5). (Error bars denote standard deviation of site co-ordinates). Arrows indicate direction of environmental variables significantly correlated with axis scores from Table 3.4

Table 3.4: Coefficients for the correlation between environmental variables with the first two axes of a canonical discriminant analysis

Environmental variable	Correlations			
	Can. 1 (74%)		Can. 2 (26%)	
Easting	0.189	*	-0.095	
Northing	0.327	***	0.081	
Distance to tidal	0.011		-0.397	***
Stream order	-0.308	***	0.098	
Altitude	0.218	*	-0.047	
Barrier downstream	0.816	***	0.088	
Sand	-0.257	**	-0.165	
Mudsilt	0.026		-0.301	**
Riffle	0.037		0.229	*
Glide	0.023		-0.265	**
Instream cover	0.195	*	-0.144	
Shade	0.040		0.284	**
Mean wetted width	-0.269	**	0.052	
Geology	-0.253	**	-0.121	
Alkalinity	-0.021		-0.297	**
Total hardness	-0.054		-0.406	***
Conductivity	-0.026		-0.314	***
Velocity rating	-0.176		-0.232	*
Fencing (lhs)	0.163		-0.299	**
Bank slippage (lhs)	0.233	*	-0.095	
Fencing (rhs)	0.240	*	-0.490	***
Bank slippage (rhs)	0.155	*	-0.110	

* denotes P<0.05, **denotes P<0.01, ***denotes P<0.0001

3.3.3 *Calculation of* O/E *Ratios*

The *O/E* ratios of the predicted and observed fish faunas were calculated using all taxa with probabilities >0.5. The expected number of taxa was obtained by summing the probability values for each of the taxa from the ranked list of weighted probabilities. This process was repeated for all reference sites, to give the distribution of *O/E* ratios (Fig. 3.2).

3.3.3.1 Model Evaluation

To evaluate the ability of the model to assess the condition of a site, the non-reference sites were run through the model and *O/E* ratios calculated in the same way as the reference sites and these values were compared. The mean *O/E*-value for all reference sites was close to unity at 0.89 (SE 0.018), which suggested that the model produced unbiased estimates of the number of taxa expected to occur at a site. The mean *O/E* ratio for the test sites was however significantly lower at 0.66 (SE 0.01) (ANOVA F_{1467} = 95.6, P <0.0001).

The *O/E* values were then grouped by Q-values where these were available to determine if there was a relationship between *O/E* score and Q-score. The *O/E* scores were significantly different for the six Q-score groups (F_{5464} =

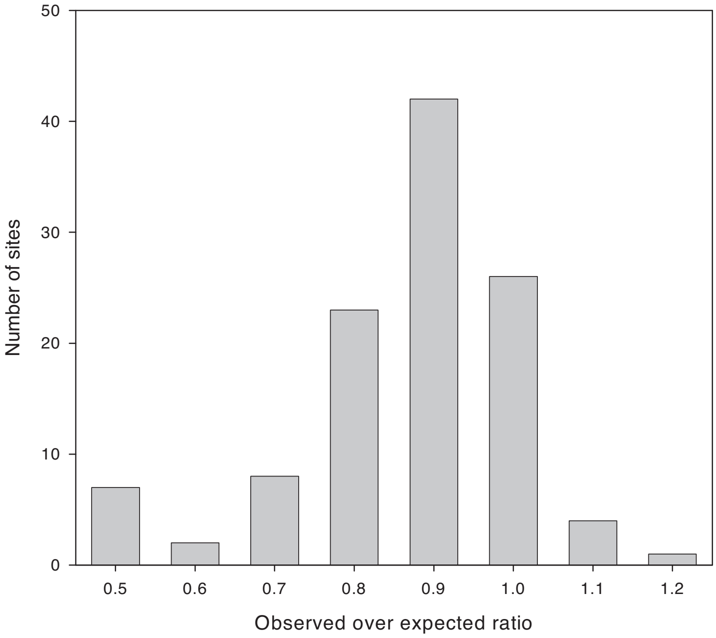

Figure 3.2: Spread of observed over expected ratios for the 114 reference sites

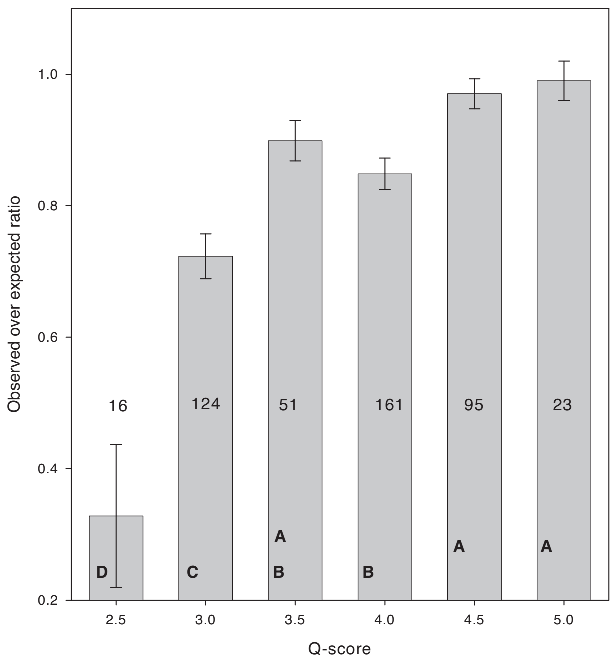

Figure 3.3: Mean and standard error for the O/E ratio scores in each Q-score class. Groups with the same letter not significantly different at the 5% level using Tukey's multiple

17.84, P<0.0001); the Tukey's mean test revealed that the differences were between the lower Q-scores but some showed that overlap occurred at the higher scores (Fig.3.3).

3.3.3.2 Probability Cut-off Levels for Inclusion of Taxa

The use of the 0.5 probability level followed the AUSRIVAS protocol rather than the more stringent 0.0 (i.e. all probabilities) level used in Britain with the RIVPACS models. Hawkins et al. (2000) found that when these two cut-off levels were compared the 0.5 cut-off level yielded more robust model outputs.

3.4 Results: Model Development of Reference Sites Without Barriers

Reference sites above barriers presented a problem as they could not be considered 'reference' where the barrier was artificial. Therefore, a new predictive model was constructed without sites above barriers. The model construction and validation followed the process in the first model described in Section 3.2 above. There were 78 sites (Q4–5 and Q5) without barriers and these were run through a TWINSPAN analysis and four groups were identified (Table 3.5). Group 1 sites contained only eels, minnows and trout. Group 2 sites contained eels, minnows, trout and salmon. Group 3 sites contained all species and Group 4 sites contained all species except salmon.

Table 3.5: Percentage of each taxon in each of the three groups

Group	1	2	3	4
Eel	40	60	70	08
Minnow	30	100	10	20
Three-spined stickleback		100	70	80
Stoneloach			90	20
Lamprey (juveniles)			60	40
Trout (total)	100	100	100	100
0+Trout	100	90	50	100
1+ & Older Trout	100	90	90	80
Salmon (total)		100	100	
0+ Salmon		90	50	
1+ & Older Salmon		90	100	

The new model without barriers showed good discrimination (an overall crossvalidated error rate of 28%) (Table 3.6) for the set of variables shown in Table 3.7.

All 470 sites including those with barriers were run through the model and the *O/E* ratios were calculated and are shown in Figure 3.4. The mean *O/E* ratios for the sites without barriers (256 sites) are shown in Figure 3.5.

Table 3.6: The crossvalidated number and percentage of sites classified correctly into each of the four TWINSPAN groups by linear discriminant analysis using the concurrently measured environmental variables listed in Table 3.7

	Predicted group membership (to group)				
Group (from group)	1	2	3	4	% of sites correctly predicted
1	**7**	1			87.5
2	2	48	1	4	87.2
3	0	3	**7**		70
4		2	1	**2**	40

Table 3.7: Variables selected for use in the second model after stepwise discriminant analysis

Variable	Partial R^2	F Value	Pr >F	Wilks' Lambda	Pr < Lambda
Stream order	0.29	9.96	<0.0001	0.71	<0.0001
Tot hardness	0.26	8.43	<0.0001	0.53	<0.0001
Easting	0.25	7.85	0.0001	t0.40	<0.0001
Fencing (rhs)	0.24	7.36	0.0002	0.30	<0.0001
Shade	0.15	4.21	0.0085	0.26	<0.0001
Sand	0.13	3.57	0.0183	0.22	<0.0001
Distance to tidal	0.14	3.57	0.0184	0.19	<0.0001
Conductivity	0.18	4.86	0.0041	0.16	<0.0001
Altitude	0.16	4.23	0.0085	0.13	<0.0001
Max depth	0.13	3.14	0.0312	0.12	<0.0001
Riffle	0.15	3.77	0.0148	0.10	<0.0001
Gravel	0.16	4.08	0.0103	0.08	<0.0001
Bank height (rhs)*	0.12	2.76	0.0496	0.07	<0.0001
Northing	0.11	2.64	0.0575	0.06	<0.0001
Water levels	0.16	3.73	0.0158	0.05	<0.0001
Bank erosion (rhs)	0.12	2.61	0.0598	0.05	<0.0001
Bank erosion (lhs)*	0.12	2.68	0.0553	0.04	<0.0001
Land use	0.11	2.31	0.0857	0.04	<0.0001
Fencing (lhs)	0.11	2.31	0.0858	0.03	<0.0001

* rhs: right-hand side of river; lhs: left-hand side of river

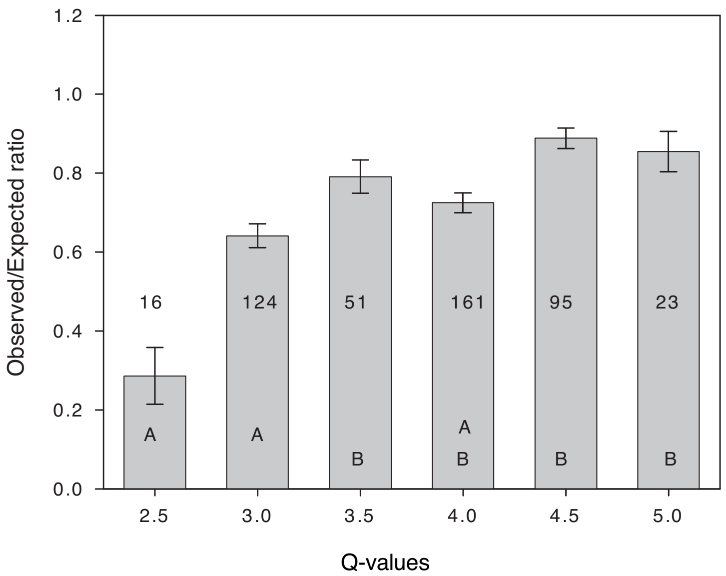

Figure 3.4: Mean O/E values and standard error for the 6 Q-value groups. (Groups with the same letter not significantly different at the 5% level using Tukey's multiple range tests, number represent the number of sites in each group)

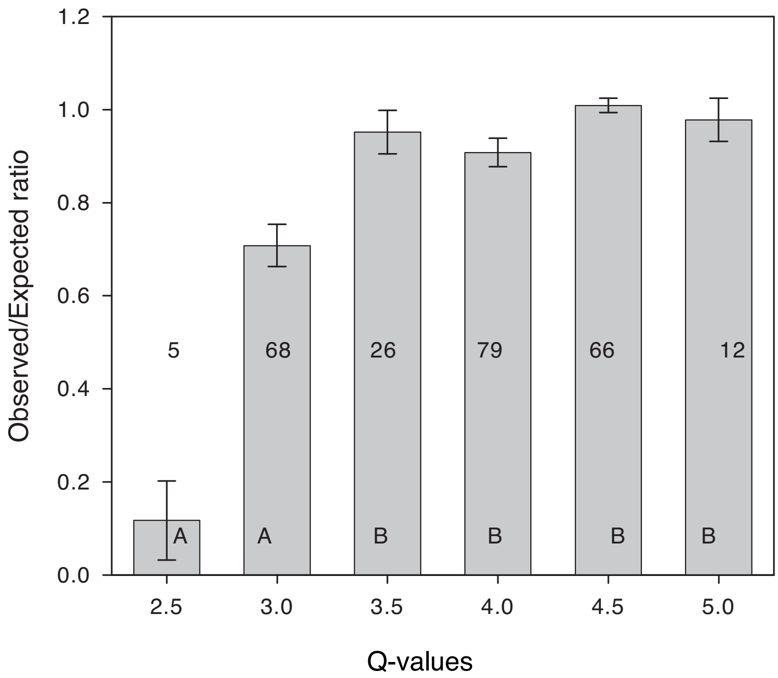

Figure 3.5: Mean *O/E* values and standard error for the 6 Q-value groups. (Groups with the same letter not significantly different at the 5% level using Tukey's multiple range tests, number represent the number of sites in each group) (without barriers)

3.5 Discussion

The 'with and without barriers' discriminant model assessment indicated the percentage of sites assigned correctly to bio-groups and showed that the distribution of reference site *O/E* scores was similar to many published RIVPACS and AUSRIVAS models produced in the UK, Australia and New Zealand using fish and invertebrates (Smith et al., 1999; Wright et al., 2000; Joy and Death, 2002; Joy and Death, 2003). This suggests that the model produced here is robust and up to the standard of other similar models in use worldwide.

The correlation with Q-scores was strong and positive but there was no significant difference between the reference sites and the Q-score 3–4 sites. There are two probable interrelated explanations for this:

- The Q-scores are calculated using invertebrate communities which are mainly influenced by local conditions and the site watershed. On the other hand, the fish are influenced by conditions upstream and downstream of the site.
- The variable which had the strongest influence on the fish communities was the presence of a barrier. This means that a number of the reference sites may be influenced by restricted fish passage, possibly negating their reference sites status (i.e. if the barrier is artificial).

Both explanations are liable to influence the results by lowering the scores of the reference sites and raising the scores of the test sites, thus causing the lack of discrimination between reference and test score.

The model constructed with high-quality sites without barriers showed a clear separation between the four TWINSPAN groups – although, with only 78 sites, some groups were small. The distribution of mean *O/E* values for all the sites after being run through the new model showed no better discrimination between Q-classes than the original model. The reason for the lack of discrimination between Q-classes, by the predictive fish model, is likely to be related to the Q-values being calculated using macroinvertebrates.

As noted above, the assessment of sites using macroinvertebrates and fish are likely to be different due to differences in the scale of environmental influences, firstly because invertebrates are influenced primarily at the proximal or reach scale and secondly by catchment process upstream of the site, while fish communities are the product of these influences but also those downstream as some migratory species come from the sea. As a result, the fish communities are indicators of the whole river from source to the sea, whereas the invertebrates integrate proximate and upstream influences only. Given these differences, it is not surprising that the two assessments do not agree totally.

The use of non-biological criteria for defining reference sites would largely circumvent the problems outlined above and allow fish communities for site assessment at the whole-river scale. Furthermore, because only presence/absence data for species were used, influences on fish abundance would not be detected in this process.

3.6 Recommendations

- Further research is required to positively identify a large number of reference sites on the basis of fish and macroinvertebrates upstream and downstream of natural and artificial barriers.
- The expanded dataset might allow more comprehensive analyses using AI and predictive modelling.

4 Development of an Appropriate Sampling Protocol for Assessment of the Status of Fish Stocks in Rivers for the WFD (and Preliminary Proposals for Lakes)

4.1 Rivers

4.1.1 Introduction

Assessing animal populations requires a sampling strategy that balances the level of accuracy and precision with the level of sampling effort in terms of personnel and cost (Southwood and Henderson, 2000). The strategy adopted varies with the nature of the investigation, the target organisms and the importance (scientific, economic, political or otherwise) of the results. Cost-effective sampling needs to match the likely error in the resulting data to the costs involved. Such uncertainty in any population assessment can occur both as a result of inherent spatial and temporal variation in the target population and the error in catching/observing and counting individuals within a given area. Both sources of error must be taken into account when designing a sampling strategy.

The need to assess fish populations in rivers has increased in importance over the years as pressures on river ecosystems have grown, particularly since the middle of the twentieth century. The Water Framework Directive (CEC, 2000) requires member states to establish monitoring programmes in accordance with Annex V and to conduct a comprehensive overview of water status in each river basin district (RBD). The WFD requires the inclusion of fish as a biological element in the assessment of ecological quality. European member states are required to establish methods and tools for assessing ecological status and guidance on the approach to classification is provided (ECOSTAT, 2003). Currently, there is no national monitoring programme for assessing the ecological status of fish in Ireland.

Electric fishing is the most widely used technique to assess fish populations in rivers and may involve one or more (multi-pass or depletion) fishings of a stretch (Zippin, 1956; Cowx, 1995). However, the process is costly in terms of staffing, equipment and time (Crisp, 2000). Costs are related to the length and area of stretch fished and the particular methods used. Quantitative methods are more costly than semi-quantitative methods.

As Section 2 showed, salmonid populations typically show high spatial and temporal variation within and between rivers. High variation in recruitment success, leading to large inter-annual population fluctuations and seasonal movements of juveniles and adults are the two major sources of temporal variation of populations within a system (Elliot, 1994; Crisp, 2000; Armstrong et al., 2003). Spatial variation in salmonid populations can be apparent at several scales. Small-scale differences in the physical structure of benthic habitat can account for large changes in salmonid abundance and community composition within a single stretch. Fish populations can vary between rivers within a catchment because of differences in gradient, stream channel dimensions, spawning accessibility, water quality and inter-specific competition (Crisp, 2000; Armstrong et al., 2003). Sampling salmonid populations accurately and precisely, given this spatial and temporal variation, is difficult. Until recently, most non-salmonid fish species were regarded as non-migratory and considered to be in static populations with their locations in rivers defined by habitat preferences at reach scales, leading to classic longitudinal zonation patterns (Huet, 1959; Lucas et al., 1998). However, many lotic fish species display strong small- and large-scale migration patterns (Wooton, 1992; Lucas et al., 1998).

Although the spatio-temporal variation of fish populations is now becoming better understood, little information exists about how to sample such populations accurately and precisely in a given river. The guidance on sample site selection given in Annex V of the WFD reveals little about the appropriate distribution or density of sites to be sampled (Irvine et al., 2002). Sampling frequencies must be sufficient to satisfy the objectives of monitoring but not place unnecessary demands on scarce and costly technical resources (Irvine et al., 2002). There are no recommendations on the number of sites to survey on any river reach.

This element of the project is an attempt to assess the degree of fish-population sampling error in two rivers, one with little longitudinal physical or chemical variation over the study length and the other with marked longitudinal physical and chemical changes, and to describe the most appropriate sampling strategy for fish in southern Irish rivers.

The aim of this work package is to develop an appropriate sampling protocol for the assessment of fish stocks in relation to water quality in rivers. The development of a robust river-fish monitoring protocol requires an assessment of the degree of variation in fish populations between different stretches of a river in relation to physical-habitat variables. Statistical assessment is then possible of the minimum number of sample stretches (or the minimum length of river) needed (for appropriate sampling of the fish populations) in order to deliver an accurate representation of the fish stock within the river.

4.1.2 Methods

Two rivers were selected for this aspect of the study. The River Douglas is an unpolluted headwater tributary of the River Araglin (Blackwater catchment), which exhibits little longitudinal change in water chemistry along its length. The Curraheen river, a small tributary of the River Lee, rising to the south of Cork city, is subject to inputs of domestic and agricultural wastes as it flows into the Lee. Water chemistry, therefore, changes considerably as it flows downstream.

Fifteen 30m stretches (each separated by at least 30m) were sampled consecutively during the summer of 2001 on each river. The standard methodology for the project included fish-stock assessment using electric fishing, hydrochemical analysis and physical/habitat survey methodologies. Three 2-minute kick samples of macroinvertebrate communities were also taken in each stretch to assess overall water quality.

The relationship between physical habitat variables and salmonid abundance was explored by means of Pearson correlation analysis on log-transformed data using individual mean measurements from each stretch. The degree of variation in abiotic and biotic parameters for each river was assessed using the coefficient of variation for each parameter (standard deviation divided by mean value, for the 15 stretches for each river), expressed as a percentage. The required sample size needed to estimate (within 95% confidence limits) the density of trout and salmon, within a given accuracy (expressed as a percentage of the true mean) was calculated according to the following formula (Eckblad, 1991):

> Required sample size, n, = [(t-value)^2 (sample variance)] / [Accuracy x sample mean], where t-value is the value taken from the t-distribution with a specified level of significance.

The value of estimating fish abundances with single-pass electric fishing was assessed by correlating the densities of salmonids estimated from the first electric fishing pass with densities estimated from the triple-pass depletion method, for each stretch.

4.1.3 Results

The Douglas had an overall greater percentage of riffle per stretch than the Curraheen and a correspondingly lower percentage of glides. The percentage of pools was similar in both rivers. Both rivers exhibited considerable change between stretches in physical habitat, but differences between stretches were more pronounced in the Curraheen (Fig.4.1). The Curraheen also exhibited much greater longitudinal physical and chemical change than the Douglas, which was reflected in water quality and macroinvertebrate communities. Trout and salmon in the Douglas were smaller than their equivalent age-classes in the Curraheen. The density and biomass of trout and salmon in the Curraheen was much more variable between stretches, with particularly high densities of 0+ trout in the upstream stretches (Fig. 4.1). Much of the variation in salmonid abundance in the two rivers can be attributed to variation in physical habitat, both locally between stretches and along the length of the river.

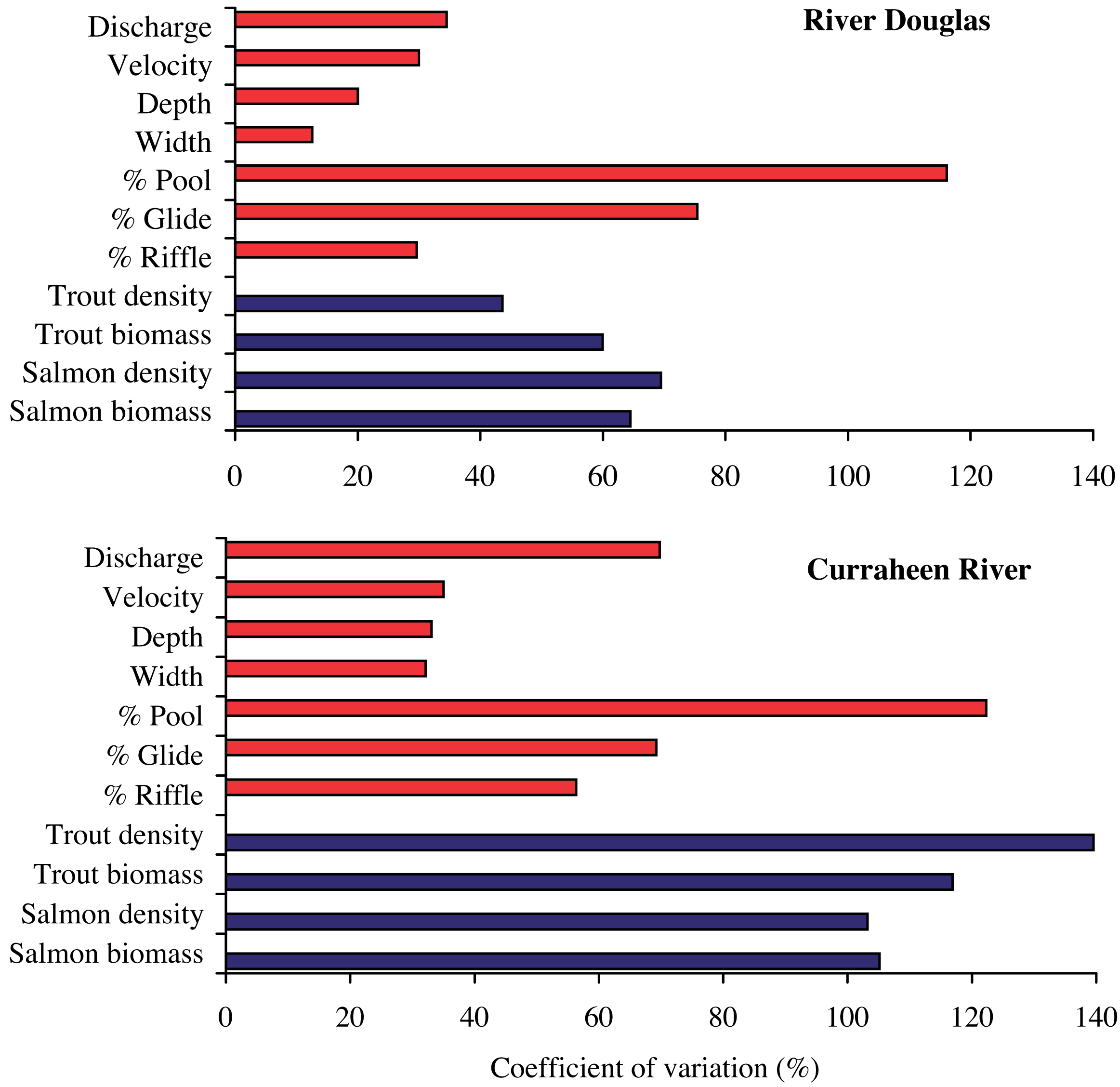

Figure 4.1: Coefficient of variation for physical habitat and salmonid density and biomass. Douglas and Curraheen River

The relationship between physical factors and salmonid abundance was markedly different between the two rivers. Trout were generally more abundant in slower flowing, deeper stretches in the Douglas, but had a strong affinity for pools in the Curraheen. Salmon showed little relationship with measured physical variables in the Douglas, but were markedly more abundant in shallow faster flowing stretches in the Curraheen (Table 4.1).

The density of salmonids caught in the first electrofishing pass for each stretch, for both rivers, was correlated against the density estimated by the Zippin multi-pass deplete fishing method (Zippin, 1956). There was a significant correlation ($P<0.05$) between single- and multi-pass estimated densities for brown trout, for all age classes, for both rivers, although the correlation between the two estimates was weaker for 0+ trout than other age groups. Correlations were also weaker for the Douglas than for the Curraheen. Similarly, the two estimates of salmon density were significantly correlated with each other for both rivers, with correlations for the Curraheen greater than for the Douglas.

The cumulative mean densities of trout and salmon were calculated from Stretches 1 to 15, for each age class and species, for both rivers. The number of cumulative sampled stretches necessary to stabilise the mean to within 10% ranged from 10–11 stretches for trout and 10–14 from salmon for the Douglas. For the Curraheen, the cumulative mean did not stabilise within the 15 stretches for trout and ranged from 11–14 stretches for salmon.

Similarly, the number of sampled stretches needed to estimate the true mean of trout and salmon was much greater in the Curraheen than the Douglas (Fig. 4.2). The number of stretches required to return an accuracy of +/- 50% for total trout density were >5 and >20 for the Douglas and Curraheen respectively and for total salmon density were 9 and 20 respectively (Fig. 4.2). Densities of fish caught in the first electric fishing pass were significantly correlated with densities estimated from triple-pass depletion methods for both trout and salmon on both rivers.

Table 4.1: Pearson correlation coefficients between salmonid and physical parameters, Douglas and Curraheen rivers

River	Physical parameter	Brown trout					Salmon			
		Total density	Total biomass	0+ density	1+ density	2+ density	Total density	Total biomass	0+ density	1+ density
Douglas	Riffle (%)	**-0.64***	**-0.71**	-0.17	**-0.80**	-0.44	0.26	-0.03	0.31	-0.38
	Glide (%)	**0.60**	**0.68**	0.19	**0.73**	0.44	-0.13	0.16	-0.17	0.38
	Pool (%)	0.33	0.36	0.04	0.45	0.20	-0.30	-0.18	-0.33	0.17
	Depth (m)	0.42	**0.63**	-0.09	**0.62**	**0.55**	-0.40	-0.24	-0.42	0.07
	Velocity (ms^{-1})	**-0.72**	**-0.61**	**-0.65**	**-0.60**	**-0.55**	-0.32	**-0.56**	-0.26	**-0.69**
Curraheen	Riffle (%)	-0.094	-0.294	0.004	-0.215	-0.408	**0.604**	**0.519**	**0.651**	-0.099
	Glide (%)	-0.134	-0.027	-0.171	-0.050	0.084	**-0.547**	-0.414	**-0.595**	0.115
	Pool (%)	**0.608**	**0.868**	0.438	**0.716**	**0.886**	-0.195	-0.316	-0.197	-0.034
	Depth (m)	-0.108	0.231	-0.220	0.024	0.488	-0.492	-0.398	-0.575	0.304
	Velocity (ms^{-1})	-0.392	-0.429	-0.303	-0.491	-0.372	**0.557**	**0.665**	0.457	**0.626**

* Significant ($P<0.05$, two-tailed test) values are indicated in bold type

4.1.4 Discussion

This investigation has shown that salmonid populations are highly variable spatially, both between neighbouring stretches along an unpolluted river and along the length of a river with considerable longitudinal chemical variation (enrichment). Much of the variation in salmonid abundance was related to physical differences (local-scale variation in mean depth, velocity and percentage pools, riffles and glides) between stretches, as found by other authors (Kennedy and Strange, 1982; Bremset and Berg, 1997; Heggenes et al., 1999 Crisp, 2000; Armstrong et al., 2003). Fish-habitat relationships differed somewhat between the two rivers. Trout were more abundant in slower flowing, deeper stretches in both rivers. Only in the Curraheen were salmon more abundant in shallower, faster-flowing stretches. When sampling salmonids, it is usually recommended that a stretch include a riffle, pool and glide, in order to standardise the physical habitat across different rivers. The strong habitat partitioning between salmon and trout in the Curraheen river emphasises the need for such a standardised approach when sampling rivers with a riffle-pool bedform.

Multi-pass depletion electric fishing, although probably the most accurate way of assessing a population at a given place and time, is nonetheless not infallible and there are several types of potential error and bias associated with

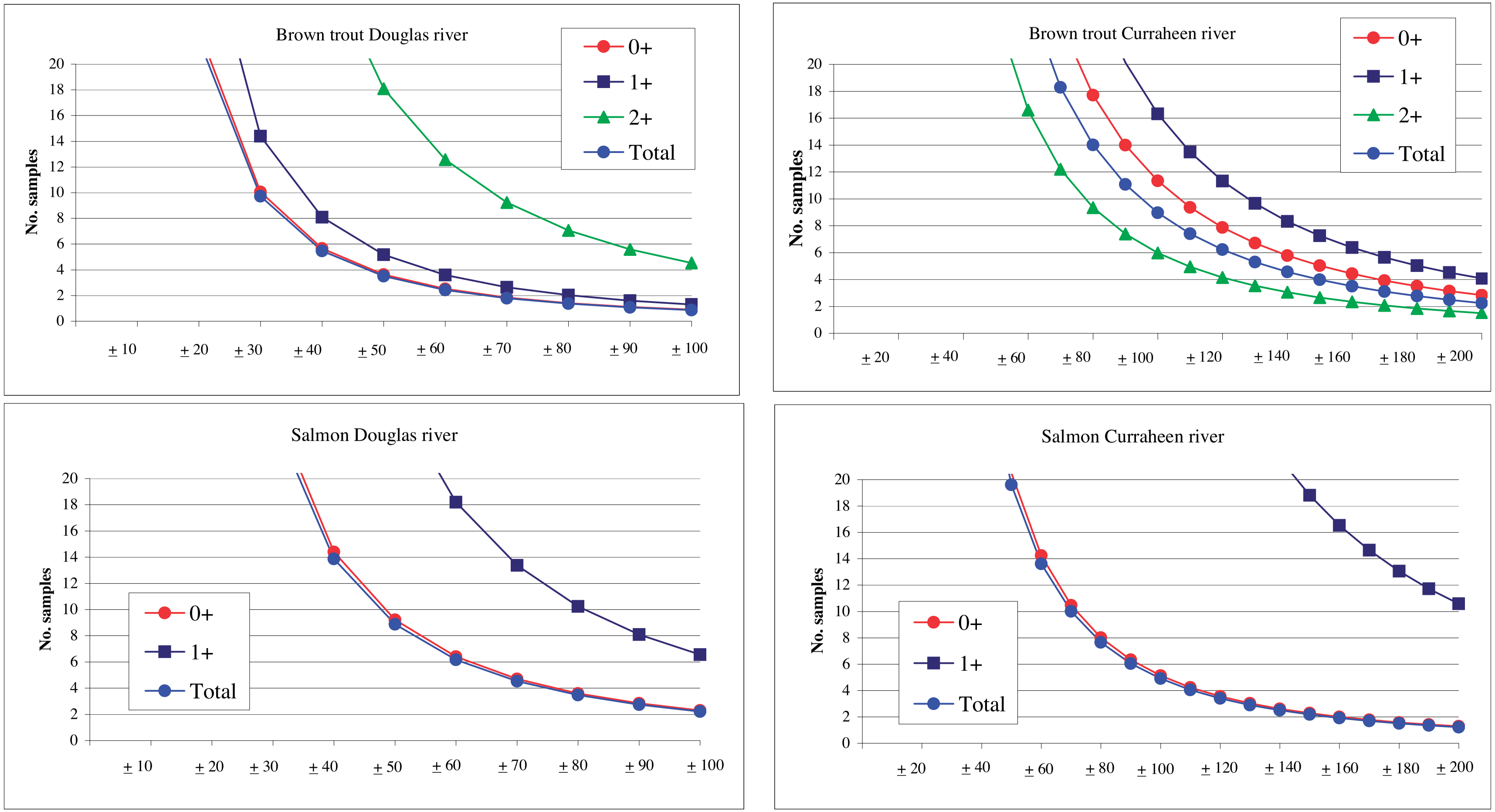

Figure 4.2: Number of samples needed to estimate (within 95% confidence limits) the density of Trout and Salmon within a given accuracy, expressed as percentage of the true mean on the Douglas and Curraheen rivers

factors such as fish size (Zalewski and Cowx, 1990), river width (Kennedy and Strange, 1982) and insensitivity to electricity (Penczak, 1985; Bohlin and Cowx, 1990). Electric-fishing efficiency can also decline between successive runs as fish become accustomed to the electric pulses (Cowx, 1995). Benthic species such as minnow, stoneloaches, sticklebacks, eels and lamprey juveniles rarely swim towards the anode (galvanotaxis) where they can be netted easily and instead become lodged under stones, boulders and vegetation, reducing their catchability strongly. Perhaps the greatest inaccuracy involved in multi-pass deplete electric fishing is through the limited stream area that can be fished for a given sample effort. Although assessing the density of fish accurately in the limited area sampled, the results should not be assumed to be indicative of the abundance of fish throughout the stream.

Single-pass electric fishing over a measured length of channel without the use of stop-nets will not catch all the fish in a given stretch. However, it has the advantage of being quicker and thus allowing a greater area to be sampled (Lobon-Cervia and Utrilla, 1993). Single-pass time-based (5 minutes) semi-quantitative assessments have been used in Northern Ireland since 1983 to monitor salmonid populations, as a practical alternative to conventional assessment methods using stop-nets (Crozier and Kennedy, 1994). Such semi-quantitative estimates are recommended where comparative data are sufficient, such as when spawning success or total abundance are compared between tributaries of a system or between successive years (Crozier and Kennedy, 1995; Wyatt, 2002). Up to 15 sites can be fished in a day using this method, allowing a great many sites to be fished in a season, and estimates to be made for recruitment from a whole river system (Crozier and Kennedy, 1995).

Mitro and Zale (2000) investigated the accuracy of single-pass electrofishing methods compared to multiple-pass methods. They found that increasing the number of sites sampled by 3 and reducing the sample effort at each site by 3 (i.e. from three-pass to single-pass) increased the precision of sampling by 48%. Double-pass methods were examined by Heimbuch et al. (1997) and were found to reduce sampling time without reducing precision greatly. Reynolds et al. (2003) found that single-pass fishing in western Oregon (USA) streams occasionally underestimated species richness by missing cryptic, mobile or rare species, but it usually estimated species richness and relative abundance as well as the intensive protocol. Importantly, single- and triple-pass population estimates tracked the same trends in population size at one site for 5 years. This current investigation found that the relationship between single-pass and multi-pass depletion density estimates varied with salmonid species, age group and stream, indicating that calibrating the 'final' density for a stretch from single-pass estimates is likely to lead to further inaccuracies.

Population estimates from multi-pass fishing of 30m stretches were highly variable. The coefficient of variation in salmonid abundance from such stretches ranged between 40% and 70% for the Douglas and between 100% and 140% for the Curraheen. Stability of means (where successive cumulative means differed by <10%) was achieved with a minimum of 10 stretches (300m) in the Douglas and not achieved at all for trout in the Curraheen. Lastly, it was found that an estimate of the mean, accurate to within 50%, would be attained from between 5 and 10 sample stretches (i.e. between 150 and 300m length of river) on the Douglas and >20 samples for the Curraheen (>600m). Clearly, a minimum length of 30m in a stream measuring 4 to 5m in width is insufficient to adequately sample salmonid densities in a river, either in one with little longitudinal change or one with large change in abiotic, physico-chemical and biological status. CEN (2003) recommends that a sufficient number of sites should be surveyed, survey lengths should be 10 to 20 times the river width and should include at least one pool, riffle or glide sequence.

Intensive multi-pass fishing may provide a good population estimate for short stretches (30m), but results suggest that this method is unlikely to be representative of the river as a whole. Assuming a maximum rate of fishing of 3 30m stretches per day, 5–10 sample stretches (giving an accuracy of not more than 50%) would take between 2 and 3 days. Such a sampling regime would be both extremely costly and high in work hours and allow for relatively few rivers to be sampled, markedly reducing the scale and extent of any monitoring programme.

4.1.5 Conclusions

1 Salmonid populations were highly variable between sample stretches, both as a result of between-stretch differences in physical habitat (both rivers) and as a result of directional longitudinal environmental change along the river (Curraheen only). The coefficient of variation in salmonid abundance from the 30m sample stretches ranged between 40% and 70% for the Douglas and between 100% and 140% for the Curraheen.

2 An estimate of the mean salmonid density, accurate to within 50%, would be attained by sampling between 5 and 10 stretches (i.e. between 150 and 300m length of river) on the Douglas and greater than 20 stretches on the Curraheen (>600m).

3 The number of sample stretches in the Douglas needed to stabilise the mean density (i.e. when successive cumulative means varied by less than 10%) was between 10 and 11 for trout and 12 to 14 for salmon. For the Curraheen, the mean density of trout failed to stabilise over the 15 sample stretches, owing to the large longitudinal variation in abundance. The mean of salmon density and biomass stabilised at between 11 and 14 sample stretches.

4 Using single or multi-pass depletion fishing in unpolluted streams <5.0m mean width, at least 300m of river would, therefore, need to be sampled to obtain a good estimate of the salmonid populations in a river with only small longitudinal variation. In a river with greater longitudinal variation such as one with a physical habitat and/or pollution gradient along its length, e.g. the Curraheen. Survey sites representing each zone (each water body type) must be sampled.

5 Densities of salmon and trout estimated from single-pass fishing correlated significantly with densities estimated by the Zippin multi-pass depletion method. Single-pass fishing is less costly and may be adequate for assessment of salmonids but is considered to be inadequate for WFD purposes where all species must be monitored.

4.1.6 Recommendations

1 Single-pass, semi-quantitative electric fishing would be a more cost-effective method than multi-pass depletion fishing for assessing salmonid populations over a large number of different rivers. However, multi-pass is the preferred option as it reduces the risk of missing rare species and possible misclassification of sites in the context of the WFD.

2 A sample length of between 300 and 450m (i.e. between 10 and 15 sample stretches of 30 to 45m in length is ideally required) would be sufficient for a typical unpolluted gravel bed river with a mean width of 5m. Such a length would also encompass several different riffle/glide/pool sequences along the river, thus reducing the error obtained from sampling a single pool or riffle of unusual physical dimensions or nature.

3 It is recommended that the sampling procedure use a standardised sample length, proportional to its width. According to CEN (2003), sampling sites must be a minimum of 10 to 20 times the river width. The length fished needs to be a balance between the degree of accuracy and precision of results required and logistical constraints such as personnel availability, time costs, accessibility of the stretch to be fished and other practical considerations. Further trials may be necessary to determine the optimal length.

4 For rivers with a high level of longitudinal variation in fish abundance (such as that due to pollution), physical habitat variation or human disturbance, then a greater sampling effort is required to obtain representative estimates of fish community composition and density.

4.2 Lakes

4.2.1 Introduction

In addition to rivers, Annex V of the WFD specifies a 3-year sampling frequency for fish stocks in lakes and transitional waters. Member states will be required to investigate fish communities (composition, abundance and age structures). Existing techniques to survey fish stocks in lakes in Ireland involve netting surveys and catch per unit effort (CPUE) estimates. A semi-quantitative sampling technique, using gill nets of different mesh size arranged in gangs (multi-panel nets) was developed by the Central Fisheries Board (O'Grady, 1981) to assess trout stocks on selected lake fisheries. Other species are also captured, as by-catch, during these surveys which have proved to be an effective management tool in illustrating the fluctuations in fish stocks over time (Delanty and O'Grady, 2001). A specialised multi-meshed monofilament survey net is used in Sweden for fish -stock assessment and the technique is being assessed for adoption as a standard for the WFD (Appelberg, 2000). However, both these methods are labour intensive and can result in high fish mortalities. Their use in privately owned lakes, or waters with rare, endangered (e.g. char) or designated (e.g. shad) species is likely to be considered unacceptable. Development of 'non-destructive' methods for fish-stock assessment is therefore required. Assessment of fish stocks in many lakes has employed hydro-acoustics (Duncan et al., 1998; Baroudy and Elliott, 1993; MacLennan and Holliday, 1996; Hughes and Hateley, 2002). Hydro-acoustics is the use of high frequency sound to measure the densities, spatial distributions and sizes of fish.

Hydro-acoustic methods for sampling fish in very deep waters are well established (Simmonds and MacLennan, 1996) and are becoming more common in shallow waters (Kubecka et al., 1994). Few studies, however, have tackled the problems of assessing fish stocks adequately in water bodies that have intermediate depths and extensive shallow littoral waters (Hughes and Hately, 2002). Both techniques (gill netting and hydro-acoustics) were compared on two lakes to provide preliminary recommendations for sampling fish in lakes, which is a requirement of this project.

4.2.2 Methods

4.2.2.1 Study Area

Two lakes, one shallow (Lough Sheelin) and one deep (Lough Melvin) were chosen for this study.

4.2.2.2 Gill Netting

The method involves the use of standard 'gangs' of gill nets ranging in mesh size from 5–12.5cm stretched mesh increasing at 1.25cm intervals. Each gang is composed of seven individual gill nets of 27.5m in length (O'Grady, 1981) joined end to end to create a single unit. These gangs were designed to capture brown trout of 19.8cm and greater. The netting sites were chosen by dividing the lakes into a numbered grid system of squares (250 x 250m), using an Ordnance Survey map (1:50,000). The specific sites were then chosen by random number generation. The nets were set at random at 30 sampling sites and fished overnight (Delanty and O'Grady, 2001).

The netting surveys were completed over 3 days on each lake (July 2001–Lough Melvin and March 2002–Lough Sheelin) using three boat crews each servicing five gangs of nets on 2 consecutive nights.

4.2.2.3 Hydro-Acoustics

The hydro-acoustic equipment was a Simrad EY 500 split-beam sonar operating at 120khz with the variable ping rate set at 10 and an interval of 0.1 seconds. Pulse duration was set to 'medium' and bandwidth to its 'wide' setting. Raw data were gathered and replayed for analysis, displayed at TVG (time varied gain) set to 40logR for single fish detection as tracks. A circular 7° beam width transducer was mounted on a pole and bracket housing and suspended 0.5m from the side of the boat and 0.4m below the water surface. A horizontally beamed elliptical (2.5 x 10° beam) transducer was tested on Lough Melvin and Lough Sheelin. Single echo detection was carried out with threshold of -70db in Lough Sheelin and -50db in Lough Melvin.

Each lake was divided into regions of approximately 1km^2, each containing at least 2km of sonar track. The length of each analysed segment varied from 150m to 350m depending on the depth. The total insonified area can be cumulatively estimated in each square from the segments. This enables estimation of the total number of fish targets per hectare for each analysis square.

Vertical sonar is not considered to be a viable method for the larger lowland lakes with extensive areas of depths less than 15m (Hughes, pers. comm.). Therefore, a horizontal transducer was used on the Lough Sheelin survey.

4.2.3 Results

4.2.3.1 Gill Netting

The principal fish species in Lough Melvin were trout, rudd, perch, char and salmon. The main fish species present in Lough Sheelin were trout, perch, roach, pike, bream and roach/bream hybrids.

4.2.3.2 Hydro-Acoustics on Lough Melvin

Numbers of fish per hectare for each analysis square ranged from zero fish to almost 200 per hectare. The average fish density was 56 individuals per hectare. Several shallow water ‘boxes’ had a count of zero, where it is known that nets caught significant numbers of fish.

4.2.3.3 Hydro-Acoustics on Lough Sheelin

The fish stock survey on Lough Sheelin was conducted from 12–14 March 2002. Unfortunately, strong winds militated against the gathering of meaningful acoustic data. Some acoustic survey tracks were attempted, using vertical pointing beams and the same SIMRAD EY500 and 7° beamwidth split-beam transducer as used in Lough Melvin. Horizontal beaming, the only practicable means of surveying such a shallow lake, resulted in echo traces which were completely obscured by noise from bubbles (Fig. 4.3).

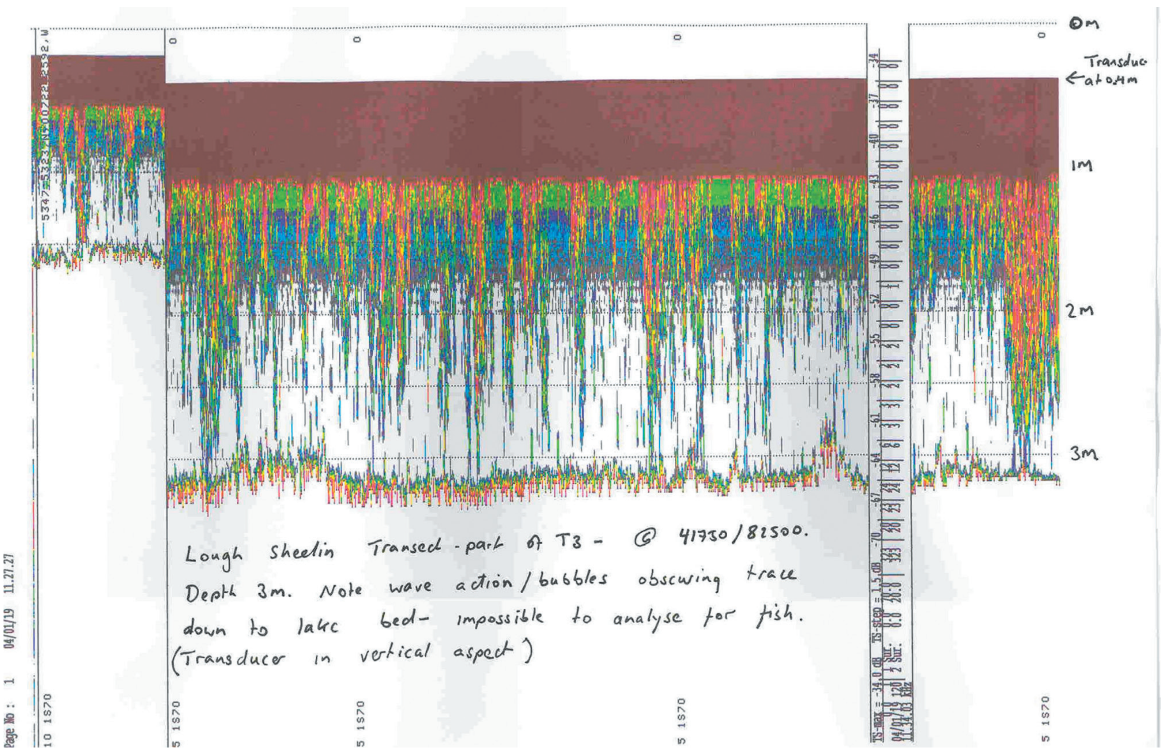

Figure 4.3: Part of echogram from shallow water in Lough Sheelin, 12 March 2003, showing entrained bubbles in shallow water

4.2.4 Discussion/Conclusions

The hydro-acoustic technique requires the use of selective deployment of gill nets to sample and therefore ‘ground truth’ the target species encountered by this technique. The intention of running both methods in tandem in this study was that the gill nets deployed would deliver the ground truth element of the acoustic survey, thereby eliminating the need for an additional ‘targeted’ netting effort. However, the traditional CFB netting survey technique, based on random selection of the netting sites, did not facilitate direct comparison of the methods. Given a short sampling period, deep water, and a structured approach to the netting, it should be possible to correlate fish in nets with sonar counts. Ideally, one would set nets in each of the analysis boxes and design the sonar track and netting survey together.

Storm conditions and excessive movement of the boat affected the fishability of the nets and created sampling difficulties using the sonar. This was more pronounced with the horizontal system on Lough Sheelin. During one short period of calm conditions, it was possible to survey two-thirds of Lough Sheelin using the hydro-acoustic equipment. Hydro-acoustic surveys were conducted on both lakes in daylight hours and because scheduling of the netting survey was fixed it was not possible to be selective for calm conditions.

It is clear that in order to survey a shallow lake like Lough Sheelin and obtain meaningful fish counts, calm conditions are necessary. This applies particularly to shallow lakes, in contrast to deep lakes where wave action also entrains bubbles but to a far shallower depth in proportion to the water column depth. It is also clear that, given the occasional very high numbers of extremely small targets, presumed to be small roach, the standard gill-netting mesh range of the trout survey method are operating as intended (net gangs designed to avoid capturing small fish) and the inclusion of smaller mesh nets would be necessary to sample the full range of age classes as required by the WFD.

4.2.5 Recommendations

1 Test acoustic estimates with nets using a structured rather than a random distribution of nets, and design the sonar track and netting survey together.

2 Calm conditions are necessary for hydro-acoustic surveys.

3 Ideally, conduct hydro-acoustic surveys during early morning or at night to avoid wind and wave action and when fish are more evenly distributed through the water column (to avoid shoaling fish) (Hughes, pers. comm.). However, this may have significant health and safety implications and requires further consideration and evaluation.

5 Microdistribution of Brown Trout and Eels in Relation to Diurnal Oxygen Variation

5.1 Introduction

Dissolved oxygen (DO) is essential and in some cases even the limiting factor for maintaining aquatic life: its depletion in water is probably the most frequent general result of certain forms of water pollution (Alabaster and Lloyd, 1980). High DO concentrations (supersaturation) during the day are often associated with deoxygenation ('oxygen sag') at night time, when respiration processes replace photosynthesis (Flanagan, 1990). Many factors can cause or contribute to the deoxygenation of a riverine system, e.g. organic loading and pollution from sewage, agricultural wastes, industrial pollutants and weather-related causes (Environment Agency, 2000). In a pristine environment DO levels just below 10mg/l O_2 or 100% saturation would be expected (Mc Garrigle, 2001).

Fish obtain the oxygen they need from that dissolved in the surrounding water, absorbing it through their gills. Sensitivity to low DO concentration differs between fish species, between the various life stages (e.g. egg, juveniles and adults) and between the different life processes (e.g. feeding, growth, etc.) (Alabaster and Lloyd, 1980). Salmonid fish will begin to be affected as DO levels drop to 50%, whereas cyprinids are affected at levels around 30% (Flanagan, 1990); below these levels mortality may occur, depending on the duration of such conditions, the acclimatisation period and water temperature.

The aim of this element of the work package was to advance understanding of the movement of two common riverine fish species in relation to diel variation in a range of variables including dissolved oxygen, through radiotelemetry.

5.2 Methods

An intensive tracking study began in summer 2002 and was repeated in summer 2003 on three rivers, two moderately polluted systems (the Rye water and the Aggard) with an excessive amount of instream plant growth and a control clean-water site (the Derreen river). The programme was designed to determine whether or not brown trout moved to areas of the river with more favourable oxygen levels in response to pre-dawn oxygen sags. The method chosen to study this behaviour was radio telemetry using external attachment of VHF radio tags. Ten brown trout were tagged on the Rye water and the Derreen. Three eels and three brown trout were tagged on the Aggard stream. Tracking, using a small VHF radiotag with a portable antenna, took place at each site on consecutive days, to maintain the validity of data, over a 5-week period in July and August 2002 and 2003 on the Rye water and the Derreen. Tracking was confined to the dawn (4 a.m. to 7 a.m.) and day period (1 p.m. to 4 p.m.) when oxygen concentrations were expected to be at their lowest and highest respectively. The location of each fish on the Aggard stream was recorded hourly for 5 days.

Fish location data for the Rye water and Derreen sites were exported to GIS (ArcView 3.3) and movements were analysed using a specialist ArcView animal movement extension (Hooge and Eichenlaub, 1997). Daily reach length was recorded from two fixes, one at dawn and one during the day period, for each fish to examine if movements were related to the variation in oxygen. The Ranges V program developed by Kenward and Hodder (1996) was also used to estimate home ranges for each fish using cluster polygons. A home range may be defined as the area that an animal normally moves in. Home ranges were recorded from fixes taken throughout the study period.

Fish activity data from the Aggard river were pooled within species because of low sample sizes. Patterns in abiotic variation were examined in the Aggard river by plotting DO, temperature and light values over time (both untransformed and angular-transformed). The existence of diel cycles in these data and fish activity was examined using circular-linear correlations and Rayleigh distribution tests (Zar, 1999). The potential association between fish activity and measures of abiotic variation (light, DO and temperature) were examined using circular-linear correlations.

5.3 Results

DO concentrations showed a consistent cycle of supersaturation during the day followed by reduced concentrations at night on all rivers in 2003 (Figs 5.1–5.3.). The difference in oxygen concentration between the dawn and day was found to be statistically significant in the Rye water and the Derreen river ($P<0.001$). Temporal variation in oxygen was highest in the eutrophic Rye water (4.38mg l^{-1}, 47.6%-18.75mg l^{-1}, 166.2% DO) in comparison to the relatively low variation in the control site, the Derreen (8.02mg l^{-1}, 84%-11.46mg l^{-1}, 110% DO).

DO concentrations also showed a consistent cycle of supersaturation on the Aggard stream during the day (max. conc. = 11.7 mg l^{-1}, 116% saturation) followed by reduced concentrations at night (min. concentration = 7 mg l^{-1}, 65.9% saturation) (Fig. 5.3).

Detailed temporal and spatial variability in DO concentration on the Rye and the Derreen was examined by Aherne (2002). Three habitat types were identified as pool, riffle and glide, and oxygen measurements were taken along a transect in each habitat type. A nested ANOVA showed no significant difference in oxygen levels between habitats (riffle, pool and glide) at both sites ($P>0.05$). However, lower oxygen levels were generally found along the margins of the Rye water in the dawn period.

Mean daily reach length of trout on the Rye water (distance recorded between the fix obtained at dawn and that taken during the day) was generally <10m in 2002 with two exceptions, fishes E1 and F1 which moved 62m and 66m respectively. This was not the case in 2003 when only 4 fish moved an average of 10m per day, 4 others moved between 10 and 34m daily and the remaining 2 trout, L1 (56m) and R1 (114m), made substantially larger daily movements than the other fish. *(These values must be used as a guide only as no assumptions are made about where the fish may have moved between fixes.)*

Mean daily reach length on the Derreen was generally shorter than the Rye water with all fish moving less than 12m in 2002 and less than 19m in 2003. In 2003, the majority of trout moved <14m with only 1 fish (C2) making larger daily movements. Analysis indicated that the daily reach length travelled by each fish on the Rye and the Derreen was significantly different in 2002 (t-test; $p = 0.002$, $t = 3.095$, $df = 182$) but not in 2003.

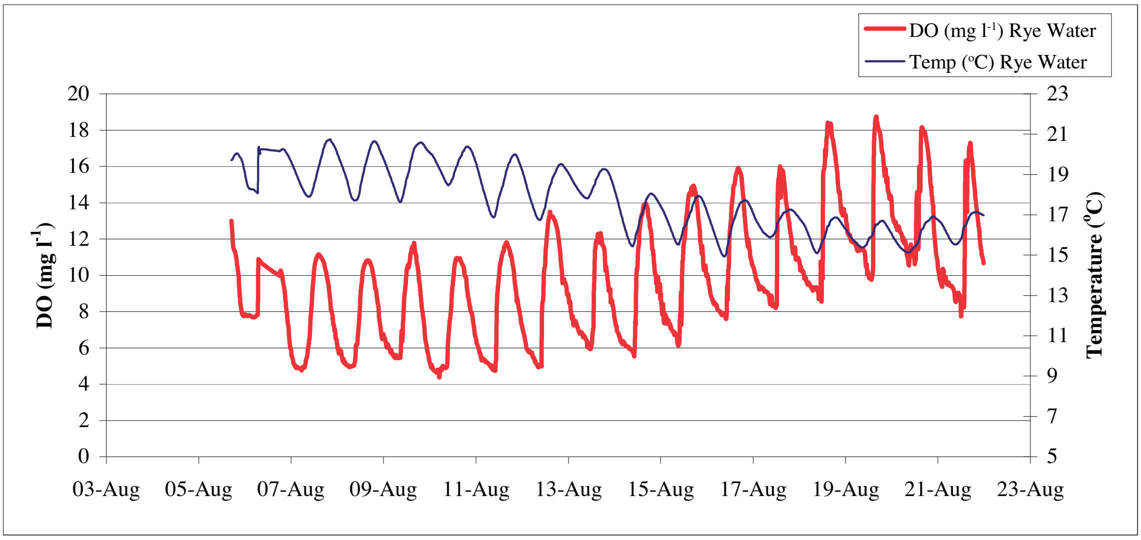

Figure 5.1: Diurnal variation in DO (mg l^{-1}) and temperature (^oC) on the Rye Water, August 2003

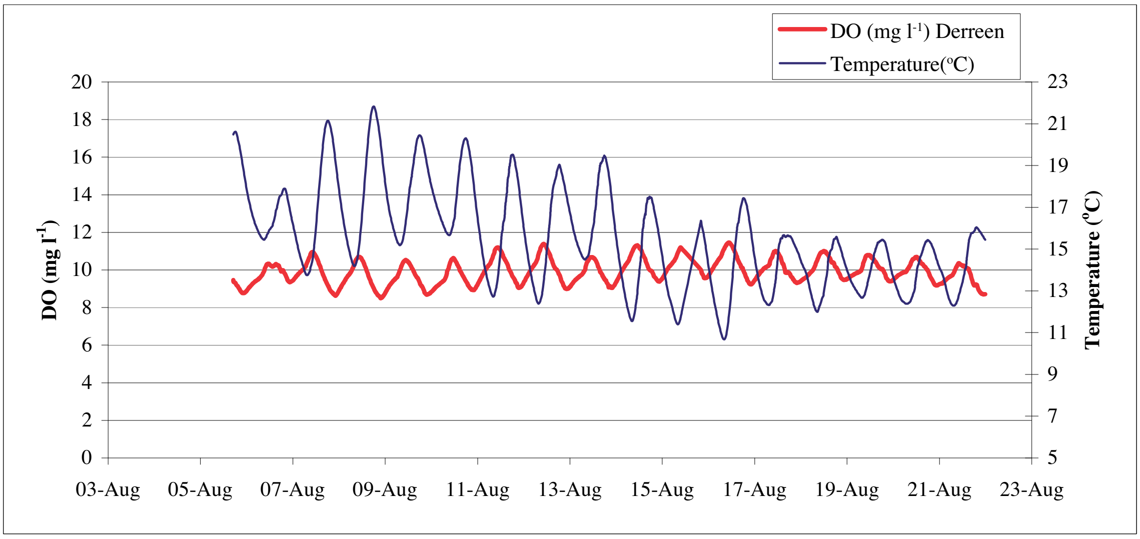

Figure 5.2: Diurnal variation in DO (mg l-1) and temperature (°C) on the Derreen river, August 2003

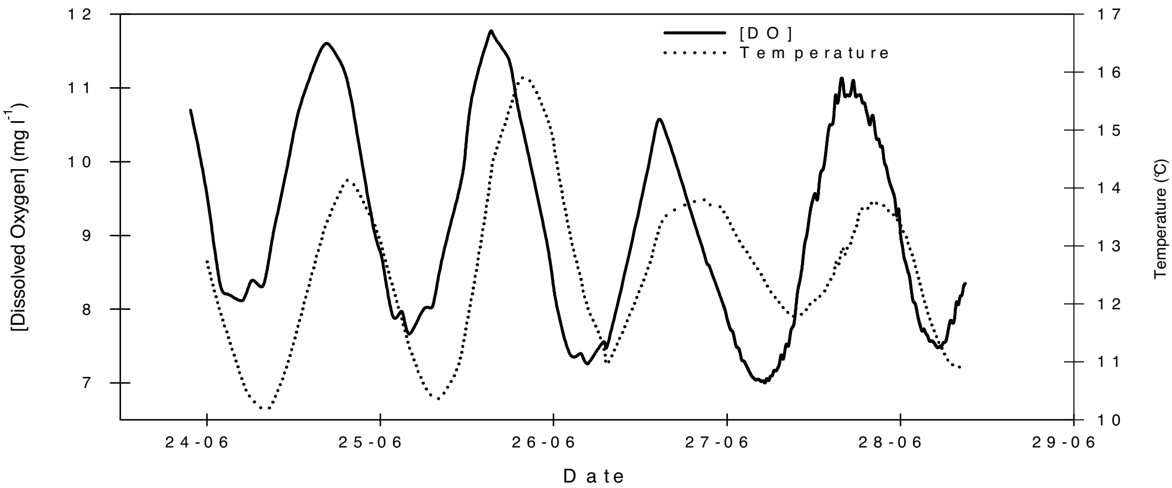

Figure 5.3: Diel variation in temperature and dissolved oxygen during period 23-28/6/03. Note significant diel shift in temperature and dissolved oxygen encountered by fish (rangetemp = 5.7°C & rangeDO = 4.7 mg l-1)

Home range was determined for fish in the Derreen and Rye water (see Table 5.1 in Kelly et al., 2007). The median value of home range (cluster polygons:fish with >38 fixes) for all trout in the Rye Water in 2003 was 295m2 which is four times as big as the median value in 2002 (68.5m2).

Home ranges used (cluster polygons) were similar between years on the Derreen where the median home range in 2002 was 86m2 and 104m2 in 2003. A Mann-Whitney U test was used to compare median values of home range (cluster polygon) using more than 38 fixes. Trout on the Rye and Derreen used similar home ranges in 2002 (Mann-Whitney U: $p = 0.789$, $z = -0.267$, $n = 14$) but they were significantly different in 2003 (Mann-Whitney U: $p = 0.046$, $z = -1.995$, $n = 16$).

The relationship between daily reach length and DO and temperature was examined on each river in each year using partial correlation. Trout movement was not correlated with DO or temperature. Even under extreme conditions when the DO variation was at its highest, daily reach movement was not statistically significantly different in 2002 or 2003 from normal conditions.

Eels on the Aggard stream typically located themselves in structurally complex refugia (e.g. amongst the branches of a submerged tree, or within macrophyte beds) and made consistent small-scale movements within a localised area. Normal eel activity (levels between 25th and 75th percentile) was not distributed equally throughout the diel cycle (Rayleigh Z = 4.9, P<0.007 (Figure 5.4a), but was concentrated weakly in the early hours of the morning (mean vector = 01:11; mean vector length = 0.25). Periods of increased eel activity (≥75th percentile, Fig. 5.4b) were not coincidental with periods of low DO (Watson's U2 = 0.422, df = 29, 45, P<0.001), or elevated water temperature (Watson's U2 = 0.64, df = 29, 45, P<0.001) which reflected findings in the Derreen river and Rye water.

On average, the tagged trout on the Aggard stream were more active, moving more often and further than eels. One individual moved more than 130m between hourly readings. However, mean movements were relatively small, and it became apparent that individual trout remained within a relatively defined area over considerable periods of time. Normal trout activity (levels between 25th and 75th percentile) was not distributed equally throughout the diel cycle (Rayleigh Z = 4.0, P<0.018), but was weakly concentrated in the early morning (mean vector = 03:59; mean vector length = 0.194). As shown above for eels, periods where trout showed increased activity (≥75th percentile) were not coincidental with periods of low DO (Watson's U2 = 1.15, df = 45, 71, P<0.001), or elevated water temperature (Watson's U2 = 0.72, df = 45, 71, P<0.001).

Trout and eels in the Aggard were found in areas with macrophyte densities different from those recorded in the environment (Kolmogorov-Smirnov tests: Eels KS = 1.93, P = 0.001; trout, KS = 1.92, P=0.001). However, comparisons of habitat use by both species during periods of low or high levels of DO or temperature showed no evidence of shifts in habitat use during extreme conditions.

5.4 Discussion

This study has demonstrated that fish inhabiting macrophyte-dominated shallow rivers can encounter marked diel fluctuations in temperature and DO during summer months. The Rye water exhibited all the symptoms of a eutrophic river. The variation in DO between dawn and day was large with oxygen levels dropping to critical levels for salmonids (50%) for short periods of time at dawn on 5 consecutive days in 2003 and rising above 120% saturation during the daytime. These fluctuations are thought to place salmonids under considerable stress and increase the likelihood of mortality. The DO concentrations did not reach critical levels (Alabaster and Lloyd, 1980) in the Derreen river and the Aggard stream.

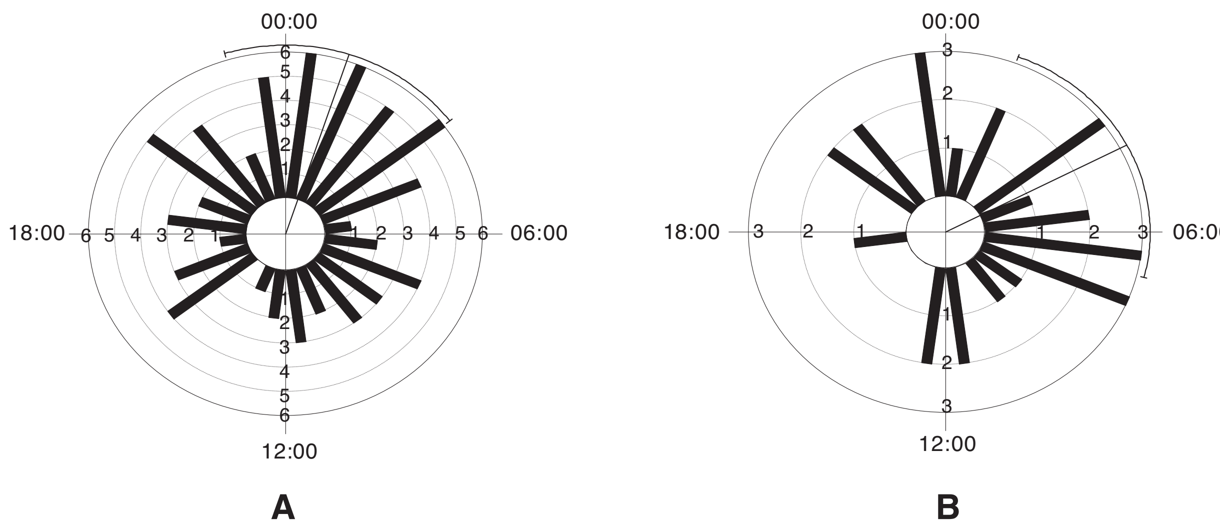

Figure 5.4: Circular histogram showing time distribution and proportions of a) medium-scale and b) larger scale activity patterns in tagged eels. NB: note use of different scale

It has been reported that during pre-dawn oxygen sags there are areas of high DO within the rivers that act as safe havens for fish (Williams et al., 2000). However, research by Aherne (2002) carried out on the Rye water (eutrophic stretch) did not find such areas. Pre-dawn oxygen sags were experienced on the Rye water in all habitat types throughout the study period. There was no conclusive evidence to suggest that brown-trout movement was related to these pre-dawn oxygen sags.

Comparison of movement data is difficult because the various authors use different definitions of home range and length of sampling section and survey period may also vary. Brown-trout movements (1+ and older) on the Afon Gwyddon, south Wales were less than 15m and movements greater than 50m were rare (Harcup et al., 1984). These movements compare well to the Derreen results. Hesthagen (1988) reported that 85–89% of brown trout aged 2+ to 9+ were recaptured within 45m of their release point. Most of them had moved less than 150m. Gowan et al. (1994) dismiss the theory that resident stream salmonids have restricted movement and suggest that salmonids exhibit large movement even during summer-feeding periods and this is a potentially common and normal phenomenon.

On the Rye in 2002, 2 fish moved further than other fish and in 2003 there were 4 such fish. Why these movements occurred is unclear: they could have resulted from oxygen variation but statistical analysis failed to link movement with DO or temperature. However, it is also possible that there was a mobile component in the population and the remaining trout were sedentary fish as outlined by Hesthagen (1988). Daily movement on the Derreen between dawn and day was considerably lower than the Rye water. Oxygen levels on the Derreen were consistently adequate so perhaps environmental conditions are less stressful to trout in this system. Daily reach movement between rivers in 2002 was significantly different but this may be explained by the difference in range of fish sizes used. Fish length ranged from 21.3cm–31.9cm on the Rye as opposed to 20cm–25.3cm on the Derreen.

From the literature, it is not possible to clarify the size of a normal home range; however, in this survey home ranges were similar to ranges found by Ovidio and Phillippart (2000). Median home ranges on the Derreen were similar in both years and broadly comparable to the Rye water in 2002. It is postulated that fish on the Rye water were subjected to greater environmental pressures with higher temperatures and wider oxygen amplitudes in 2003, which may have contributed to the larger home ranges in that year. Some individual fish in the Rye water remained stationary in marginal reed beds for extended periods. During extreme conditions, it is possible that they conserve their energy by sitting in the reeds.

Movements on the Rye water in 2003 appear to be different to those recorded in 2002 and those recorded on the Derreen. However, no statistical association was made between movement and oxygen variation or temperature even under extreme conditions. There is evidence to suggest that fish may acclimatise to low DO and high temperatures (Doudoroff and Shumway, 1970; Alabaster and Lloyd, 1980). If DO or temperature change gradually over time and remain at a critical level for a short period of time before returning to normal then fish can tolerate these stresses up to a certain lethal limit. The fish may experience a variety of physiological effects that may culminate in death if conditions remain unsuitable for a prolonged period (Davies, 1975; Doudoroff and Shumway, 1970; Alabaster and Lloyd, 1980).

Both eels and brown trout on the Aggard stream were more active during the early morning, but there was no evidence that either species encountered temperature or DO conditions that led to behavioural shifts or modified habitat use. However, the amount of variation recorded could well be expected to influence habitat choice in both species. Movements of eels in the Aggard stream were relatively limited, in marked contrast with studies conducted in a larger Irish river (McGovern and McCarthy, 1992), but those of trout were within the scope described by other workers (Young, 1999). There is a comprehensive literature describing how even subtle variation in abiotic conditions can influence the behaviour of fishes (Godin, 1997) and it was unexpected that there would be no obvious influence of oxygen and temperature on fish movement in these studies. Although this was a relatively small-scale study, it seems apparent that both brown trout and European eels are able to withstand marked fluctuations in their abiotic environment.

5.5 Conclusions

The Rye water experienced adverse environmental conditions associated with eutrophication in the form of pronounced oxygen sags at dawn and supersaturation during the day. Daily oxygen variation was larger in 2003 than in 2002 and this resulted in extreme conditions that may have caused stress to the trout populations. However, the lethal limit for brown trout was never reached. Movement in the Derreen was similar to that recorded by other authors. Fish movement was not correlated with DO and temperature but home ranges were larger on the Rye water in 2003 than in 2002. Oxygen levels did fall below 5 mg l^{-1} so fish were stressed (some remaining stationary amongst the reeds) and this may have made them more susceptible to predation in the Rye water. Trout and eels in the Aggard stream showed no evidence of increased mobility when exposed to the environmental variables experienced in the current study.

5.6 Recommendations

More research is needed in this area to:

1 Establish whether trout move at oxygen levels below 4mg l^{-1} or seek refuge in slack water where they remain stationary.

2 Determine at what oxygen concentration in the field a fish kill might occur.

3 Evaluate the effects of sub-lethal stress.

6 Survival of Salmonid Eggs along a Water Quality/ Nutrient Gradient

6.1 Introduction

The brown-trout spawning habitat is extremely important because it controls fish population dynamics within streams (Elliott, 1994). The proliferation of plant growth caused by nutrient enrichment has been associated with siltation of spawning gravels and reduction in DO, which has a detrimental effect on egg/alevin survival (O'Connor, 1998; Soulsby et al., 2001).

Salmonids preferentially select clean, well-oxygenated gravels when spawning, as fines may affect egg survival. Fines are usually defined as particles <1mm, although some authors refer to fines as any particle <2mm (Ottaway and Carling, 1981; Soulsby et al., 2001). The fine particles (<1mm) may become entrapped in small gravel thus reducing essential intergravel flow as it supplies DO to the eggs and removes metabolic wastes from the developing alevins (Hausle and Coble, 1976). Fine sediments can also adhere to and abrade the chorion of salmonid ova (Adams and Beschta, 1980). Furthermore, accumulation of fines material may trap the alevin and reduce their living space and escape cover (Dill and Northcote, 1970).

Oxygen is essential for embryo survival and is obtained as water moves through the redd. DO concentration is determined by the flow through the gravel, which in turn is affected by gravel-bed porosity (Crisp, 1993). Hatching success and alevin size are strongly dependent on gravel permeability and concentrations of DO (Turnpenny and Williams, 1980). Low oxygen concentration can also result in a reduction in fry size (Kondolf and Wolman, 1993).

Although there has been extensive research carried out on brown trout egg and alevin survival rates in relation to fines and DO concentration throughout Europe, little work has been carried out to date on linking these factors to eutrophication and examining survival along a water quality/nutrient gradient.

The main aim of this work package was to examine the survival of eyed ova and alevin, incubated in hatching boxes in river sites representing a water quality gradient on the east and west coast of Ireland, and to examine which biological, chemical or physical factors were affecting survival.

6.2 Methods

A study was carried out in 2003 on 26 sites, ranging from Q2–3 to Q5. Fifty brown-trout eggs were placed with washed gravel in 18 vibert boxes. The 18 vibert boxes were placed in three groups of six boxes and buried 10cm below the gravel at each site. Egg boxes were lifted at weekly intervals over the 6-week period and mortality was determined.

Average mortality was then calculated for each Q-value band. Mortality was determined for four different life stages: (i) egg, (ii) hatching, (iii) emergent and (iv) swim-up. However, there may have been some overlap between development stages at each site.

Logistic multiple regression using a stepwise progression was carried out to show how much of the variation in mortality was accounted for by using the relationship between the 'variables' and the 'effect', i.e. mortality. Analysis was carried out on the logit scale.

6.3 Results

Mortality increased over time but the pattern differed between site and development stage. Egg mortality was low across all Q-values. Mortality reached its highest level in the Q4–5 band largely because of two sites. These low values were attributed to hydrochemical conditions. Hatching in 2003 showed little variation across sites. Mortality during the emergent stage was highest at the Q3–4 sites. One of the sites in this band had extremely high mortality (median 96%), which lowered the overall band average. However, at the other sites, survival was good and mortality values remained below 60%. At the end of the study period, mortality at the swim-up stage was similar between sites rated Q3, Q4 and Q4–5. High mortality was found again at Q3–4 and at Q5. The lowest mortality was at the most polluted sites, which were in Q2–3 quality band. These sites had the highest survival throughout the survey.

A logistic regression was carried out using a stepwise procedure to examine which factors were having an effect on mortality. All data from sites in the east and west were combined to produce a model (Table 6.1). The following parameters – fines<2mm, pH, MRP, temperature and ammonia – met the 0.15 significant level for entry into the model and therefore in combination were having an effect on egg /alevin mortality.

6.4 Discussion

There was no statistical difference between survival rates in the various water-quality bands: survival was consistently higher at the low Q-value sites at each stage of development. Egg survival was >90% at all sites with the exception of two Q4–5 sites; one had high levels of unionised ammonia during sampling and the other was a high-altitude site in the Wicklow Mountains, which may have been subjected to acid pulses (Kelly-Quinn, pers. comm.). Egg survival of 91% was also found by other authors using similar methodology (Harshbarger and Porter, 1982).

Survival at hatching was >43% at all sites. This value was similar to survival rates of brook trout (40.6% when fines <0.85 were greater than 25%) (Argent and Flebbe, 1999). The highest hatching survival was found at the Q4–5 sites (85%), which is similar to hatching success of brown trout in a control stretch in south Wales (Turnpenny and Williams, 1980). Survival to emergence was lowest at Q3–4 sites. This band included one site that had consistently poor survival throughout the survey period.

The highest survival rates at the swim-up stage were at the Q2–3 sites. Intermediate survival rates were recorded at the Q3, Q4 and Q4–5 sites and was similar to that recorded in previous studies for direct planting of brown trout eggs. The lowest survival rates were recorded at Q5 and Q3–4 sites. The survival recorded at Q3–4 was identical to that recorded for eggs planted by Harshbarger and Porter (1982). The survival at Q2–3 sites was substantially higher than survival figures reported in the literature. It was similar to the finding of Hausle and Coble (1976).

The logistic regression model identified five main factors which affected egg/alevin mortality – (i) MRP, (ii) fines <2mm, (iii) pH, (iv) ammonia and (v) temperature. The first factor selected by the model was MRP. Although it does not have a direct effect on biota, high concentrations indicate anthropogenic inputs and levels >30ug/l are indicative of eutrophic conditions (McGarrigle, 2001).

Fines <2mm had an effect on mortality by clogging spawning gravel, and thereby reducing the DO available to the eggs/alevins. O'Connor (1998) determined that 100% mortality of alevins occurred when incubated with 25% fines (0.063–1mm), and concluded that oxygen deprivation because of clogging of the interstitial pockets caused the mortality, and that infiltration of >10% could have a negative effect on alevin survival.

Table 6.1: Model for all data in 2003 combined (0.15 = significant level for entry into the model)

	Parameter Estimator	Standard error	Type II SS	F value	Pr>F
Intercept	-7.63	2.64	30.80	8.33	0.0049
Fines (2mm)	0.10	0.03	38.12	10.30	0.0018
pH	0.68	0.34	14.29	Variable	0.0525
MRP*	-14.89	4.61	38.63	10.44	0.0017
Temperature	0.36	0.10	44.55	12.04	0.0008
Ammonia	2.70	1.09	22.70	6.14	0.0151

*MRP = molybdate reactive phosphorus

River pH was the third factor that affected egg/alevin mortality. The relationship between pH and egg mortality is well established (Weatherley et al., 1990; Kelly-Quinn et al., 1993; Donaghy and Verspoor, 1997). Research carried out in Wicklow Mountain streams found that eggs became brittle at sites exposed to acid pulses especially when pH fell below 4.5 (Kelly-Quinn et al., 1993). The pH range suitable for fisheries is between 6.5 and 8.5 (Flanagan, 1990). In this study, pH values often plummeted to less than pH 4, especially in the mountainous regions (clean sites Q4–5 and Q5), causing high egg mortality.

Ammonia was another factor isolated by the model. Total ammonia in surface waters is usually less than 0.2mg/l N but may reach 2–3mg/l N; however, these levels are indicative of organic pollution. Levels >0.2 mg/l N were only exceeded at two sites.

Water temperature was the final important parameter highlighted in the logistic model. Temperature throughout the study was within the optimum range 0–13°C for brown-trout eggs (Elliott, 1981). Hatching of 50% of brown trout occurred between 437 and 480 degree days. Eggs laid in November or December will usually hatch after 444–degree days (Elliott, 1994) and so the present results were within the expected range.

6.5 Conclusions

This research has highlighted the variability in egg and alevin survival rates. Observed mortality at the egg/alevin stage were site specific and did not follow the expected pattern of increasing mortality with decreasing Q-value. High survival rates found at some enriched sites suggested that conditions at these sites were tolerable to brown-trout eggs and alevins. However, this is no indication of whether the site could sustain a brown trout population throughout the entire life cycle. It was not possible in this study to isolate mortalities (eggs/alevins) associated solely with eutrophication effects.

6.6 Recommendations

1 Further research should be conducted to investigate why egg/alevin survival is good at slightly polluted sites and if this survival is a winter effect and whether it can be sustained throughout the following summer.

2 Further tests should include comprehensive monitoring of oxygen within the redds at each study location.

3 Eggs should be placed in the gravel at the earliest possible date with minimal disturbance of the gravel.

7 References

Adams, J.N. and Beschta, R.L. (1980) Gravel bed composition in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences,* **37**, 1514–21.

Aherne, B. (2002) Temporal and spatial variability in dissolved oxygen concentrations in two contrasting rivers: the Rye Water, Co. Kildare and The Derreen river, Co. Carlow. MSc Thesis National University of Ireland.

Alabaster, J.S. and Lloyd, R. (1980). *Water Quality Criteria for Freshwater Fishes*, Butterworth Scientific, London and others.

Angermeier, P.L. and Davideanu, G. (2004) Using fish communities to assess streams in Romania: initial development of an Index of Biotic Integrity. Hydrobiologia, **511(1)**, 65–78.

Angermeier, P.L. and Karr, J.D. (1986) Applying an Index of Biotic Integrity based on stream-fish communities: consideration in sampling and interpretation. *North American Journal of Fisheries Management*, **6**, 418–29.

Appelberg, M. (2000) Swedish standard methods for sampling freshwater fish with multi-mesh gillnets, *Fiskeriverket Information*, **1**, 3–32.

Appelberg, M., Bergquist, B.C. and Degerman, E. (2000) Using fish to assess environmental distrurbance of Swedish lakes and streams – a preliminary approach. *Verh. Internationale Verein. Limnologie*, **27**, 311–15.

Argent, D.G. and Flebbe, P.A. (1999) Fine sediment effects on brook trout eggs in laboratory streams. *Fisheries Research*, **39**, 253–62.

Armstrong, J.D., Kemp, P.S., Kennedy, G.J.A., Ladle, M. and Milner, N.J. (2003). Habitat requirements of Atlantic salmon and brown trout in rivers and streams. *Fisheries Research*, **62**, 143–70.

Baroudy and Elliott, J.M. (1993) The effect of large-scale spatial variation of pelagic fish on hydroacoustic estimates of their population density in Windermere (northwest England). *Ecology of Freshwater Fish*, **2(4)**, 160–6.

Belpaire, C., Smolders, R., Auweele, I.V., Ercken, D., Breine, J., van Thuyne, G. and Ollevier, F. (2000) An Index of Biotic Integrity characterizing fish populations and the ecological quality of Flandrian water bodies. *Hydrobiologia*, **434,** 17–33.

Berkman, H.E. and Rabeni, C.F. (1986) Biomonitors of stream quality in agricultural areas: Fish versus invertebrates. *Environmental Management*, **10(3)**, 413–19.

Bohlin, T. and Cowx, I.G. (1990) Implications of unequal probability of capture by electric fishing on the estimation of population size, in Cowx, I.G. and Lamarque, P. (eds) *Fishing With Electricity*, Fishing News Books, Blackwell Scientific Publications, Oxford, 145–55.

Bowman, J.J. and Clabby, K.J. (1998) Water quality of rivers and lakes in the Republic of Ireland, in Wilson, J.G.W. (ed.) *Eutrophication in Irish Waters*, Royal Irish Academy, Dublin, 55–63.

Bremset, G. and Berg, O.K. (1997). Density, size-at-age and distribution of young Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) in deep river pools. *Canadian Journal of Fisheries and Aquatic Sciences*, **54,** 2827–36.

CEC (2000) Directive 2000/60/EC of the European Parliament and of the Council: establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities*, Luxembourg, 72pp.

CEN (2003) *Water Quality – Sampling of fish with electricity*. European standard. Ref. No. EN 14011.

Champ, W.S.T. (1983) Quantification of trout stocks in an alkaline river fishery. *Irish Fisheries Investigation Series A*, **23**, 47–51.

Champ, W.S.T. (2000) Habitat quality for fishes. In: Sixteenth Annual Environmental Conference: *Water – Is its quality deteriorating*. Sherkin Island Marine Station, Sherkin Island, Co. Cork.

Clabby, K.J., Lucey, J.L.M., McGarrigle, M.L., Bowman, J., Flanagan, P.J. and Toner, P.F. (1992) *Water Quality in Ireland 1987–1990. Part One: General Assessment*, Environmental Research Unit, Dublin, 189pp.

Clabby, K.J., Lucey, J. and McGarrigle, M.L. (2001) *Interim report on the Biological Survey of River Quality Results of the 2000 Investigations*, Environmental Protection Agency, Wexford.

Clabby, K.J., Lucey, J. and McGarrigle, M.L. (2002) *Interim Report on the Biological Survey of River Quality Results of the 2001 Investigations*, Environmental Protection Agency, Wexford.

Clarke, K.R. and Gorley, R.N. (2001) Primer Version 5: User Manual tutorial, Plymouth Marine Laboratory, UK.

Clarke, R. T., Furse, M.T., Wright, J.F. and Moss, D. (1996) Derivation of a biological quality index for river sites: comparison of the observed with expected fauna. *Journal of Applied Statistics*, **23**, 311–22.

Cowx, I.G. (1995). Fish stock assessment – a biological basis for sound ecological management, in Harper, D.M. and Ferguson, A.J.D. (eds) *The Ecological basis for River Management*, John Wiley & Sons, Chichester, 375–88.

Cowx, I. G. and Welcomme, R.L. (eds) (1998) *Rehabilitation of Rivers for Fish*. A study undertaken by the European Inland Fisheries Advisory Commission of FAO, Fishing News Books, Blackwell Science, Oxford, 260pp.

Crisp, D.T. (1993) The ability of UK salmonid alevins to emerge through a sand layer. *Journal of Fish Biology,* **43,** 656–8.

Crisp, D.T. (2000). *Trout and Salmon: Ecology, Conservation and Rehabilitation*, Fishing News Books, Blackwell Scientific, Oxford.

Crozier, W.W. and Kennedy, G.J.A. (1994) Application of semiquantitative electrofishing to juvenile salmonid stock surveys. *Journal of Fish Biology*, **45**, 159–64.

Crozier, W.W. and Kennedy, G.J.A. (1995) The relationship between a summer fry (0+) index, derived from semi-quantitative electrofishing, and egg deposition of Atlantic salmon, in the River Bush, Northern Ireland. *Journal of Fish Biology*, **47**, 1055–62.

Davies, J.C. (1975) Minimum dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. *Journal of the Fisheries Research Board of Canada*, **32**, 2295–332.

Delanty, K. and O'Grady, M. (2001) *Lough Melvin Fish Stock Report*, Central Fisheries Board, unpublished report.

Dill, L.M. and Northcote, T.G. (1970) Effects of some environmental factors on survival, condition, and timing of emergence of Chum salmon fry (*Oncorhynchus kete*). *Journal of Fisheries Research Board of Canada,* **27(1)**, 196–201.

Donaghy, M.J. and Verspoor, E. (1997) Egg survival and timing of hatch in two scottish atlantic salmon stocks. *Journal of Fish Biology*, **51**, 211–14.

Donohue, I., McGarrigle, M.L. and Mills, P. (2006) Linking catchment characteristics and water chemistry with the ecological status of Irish rivers. *Water Research*, **40,** 91–8.

Doudoroff, P. and Shumway, D.L. (1970) Dissolved oxygen requirements of freshwater fishes. *FAO Fisheries Technical Report*, **86**, 291.

Duncan, A., Schmid and Kubecka, J. (1998) Aspects of spatial dispersion of coarse fish densities in lowland rivers by acoustic survey. *Verhandlungen Internationale Vereinigung für Theoretische und Angewandte Limnologie*, **27(5)**, 3015–18.

Eckblad, J.W. (1991) How many samples should be taken? *Biosciences*, **41**, 346–8.

ECOSTAT (2003) *Overall approach to the Classification of Ecological Status and Ecological Potential.* Water Framework Directive Common Implementation Strategy Working Group 2A Ecological Status (ECOSTAT).

Ekloev, A.G., Greenberg, L.A., Bronmark, C., Larsson, P. and Berglund, O. (1999) Influence of water quality, habitat and species richness on brown trout populations. *Journal of Fish Biology*, 554, 33–43.

Elliott, J.M. (1981) Some aspects of thermal stress on freshwater teleosts, in Pickering, A.D. (ed.) *Stress and Fish*, Academic Press, London, 209–45.

Elliot, J.M. (1994). *Quantitative Ecology and the Brown Trout.* Oxford University Press, Oxford.

Environment Agency (2000) Review of Eutrophication-related R&D. Technical Report P408.

FAME CONSORTIUM (2004) Manual for the application of the European Fish Index – EFI. A fish based method to assess the ecological status of European rivers in support of the water Framework Directive. Version 1.1, January 2005.

Flanagan, P.J. (1990) *Parameters of Water Quality. Interpretation and Standards*, 2nd edn, ERU, St Martin's House, Waterloo Road, Dublin 4, 153pp.

Flanagan, P.J. and Toner, P. F. (1972) *The National Survey of Irish Rivers.* A Report on Water Quality, An Foras Forbatha, Dublin.

Gerritson, J. (1995) Additive biological indices for resource management. *Journal of the North American Benthological Society,* **14**, 451–7.

Godin, J. (Ed.) (1997) *Behavioural Ecology of Teleost Fishes*, Oxford, Oxford University Press.

Gowan, C., Young, M.K., Fausch, K.D. and Riley, S.C. (1994) Restricted movement in resident stream salmonids: A lost paradigm? *Canadian Journal of Fisheries and Aquatic Sciences*, **51**, 2626–37.

Harcup, M.F., Williams, R., Ellis, D.M. (1984) Movements of brown trout, *Salmo trutta L.*, in the river Gwyddon, South Wales. *Journal of Fish Biology*, **24**, 415–26.

Harris, J.H. (1995) The use of fish in ecological assessments. *Australian Journal of Ecology*, **20**, 65–80.

Harris, J.H. and Silveira, R. (1999) Large-scale assessments of river health using an Index of Biotic Integrity with low-diversity fish communities. *Freshwater Biology,* **41**, 235–52.

Harshbarger, T.J. and Porter, P.E. (1982) Embryo survival and fry emergence from two methods of planting brown trout eggs. *North American Journal of Fisheries Management*, **2**, 84–9.

Hausle, D.A. and Coble, D.W. (1976) Influence of sand in redds on survival and emergence of brook trout (*Salvelinus fontinalis*). *Transactions of the American Fisheries Society*, **105**, 57–63.

Hawkins C.P., Norris R.H., Hogue J.N. and Feminella, J.W. (2000) Development and evaluation of predictive models for measuring the biological integrity of streams. *Ecological Applications*, **10**, 1456–77.

Heggenes, J., Bagliniere, J.L. and Cunjak, R.A. (1999) Spatial niche variability for young Atlantic salmon (*Salmo salar*) and brown trout (*S. trutta*) in heterogeneous streams. *Ecology of Freshwater Fish,* **8**, 1–21.

Heimbuch, D.G., Wilson, H.T., Weisberg, S.B., Volstad, J.J.H. and Kazyak, P.F. (1997) Estimating fish abundance in stream surveys by using double-pass removal sampling. *Transactions of the American Fisheries Society*, **126**, 795–803.

Hendry, K, Cragg-Hine, D., O'Grady, M., Sambrook, H. and Stephen, A. (2003) Management of habitat for rehabilitation and enhancement of salmonid stocks. *Fisheries Research*, **62**, 171–92.

Hestagen, T. (1988) Movement of brown trout, *Salmo trutta*, and juvenile Atlantic salmon, *Salmo salar*, in a coastal stream in northern Norway. *Journal of Fish Biology*, **32**, 639–53.

Hill, M.O. (1979) DECORANA- a FORTRAN program for detrended correspondence analysis and reciprocal averaging. Ithaca, New York, Ecology and Systematics, Cornell University.

Hill, M.O. (1994) *DECORANA and TWINSPAN, for ordination and classification of multivariate species data*, Institute of Terrestrial Ecology, England, 58pp.

Hooge, P.N., Eichenlaub, W.B. and Solomon, E.K. (1997) Animal movement extension to Arcview v2. (http://www.absc.usgs.gov/glba/gistools/)

Huet, M., (1959) Profiles and biology of western European streams as related to fish management. *Transactions of the American Fisheries Society.* **88(3),** 155–63.

Hughes, S. and Hateley, J. (2002) Hydroacoustic fish stock assessment in lakes in England and Wales, in Cowx, I. (ed.) *Management and Ecology of Lake and Reservoir Fisheries,* Blackwell Science, Oxford, 3–11.

Irvine, K., Boelens, R., Fitzsimmons, J., Kemp, A. and Johnston, P. (2002) *Review of Monitoring and Research to Meet the Needs of the EU Water Framework Directive* (2000-DS-5-M1) Final report. Environmental protection Agency, Ireland, 147pp.

Joy, M.K. and Death, R.G. (2002) Predictive modelling of freshwater fish as a biomonitoring tool in New Zealand. *Freshwater Biology*, **47**, 2261–75.

Joy, M.K. and Death, R.G. (2003) Biological assessment of rivers in the Manawatu-Wanganui region of New Zealand using a predictive macroinvertebrate model. *New Zealand Journal of Marine and Freshwater Research*, **33**, 367–79.

Karr, J.R. (1981) Assessment of biotic integrity using fish communities. *Fisheries*, **6(6)**, 21–7.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R. and Schlosser, I.J. (1986) Assessing biological integrity in running water: a method and its rationale, Illinois Natural History Survey, Special publication 5.

Karr, J.R., Yant, P.R., Fausch, K.D. and Schlosser, I.J. (1987) Spatial and temporal variability of the index of biotic integrity in three Midwestern streams. *Transactions of the American Fisheries Society*, **116**, 1–11.

Kelly, F.L. (2001) Standard Survey Methodologies. Unpublished project report, 18pp.

Kelly, F.L., Champ, T., McDonnell, N., Kelly-Quinn, Harrison, S., Arbuthnott, A., Giller, P., Joy, M., McCarthy, T.K., Cullen, P., Harrod, C., Jordan, P., Griffiths, D. and Rosell, R. (2007) *Investigation of the Relationship between Fish Stocks, Ecological Quality Ratings (Q-values), Environmental Factors and Degree of Eutrophication.* Final Report. Environmental RTDI Programme 2000–2006. Environmental Protection Agency, Wexford.

Kelly-Quinn, M., Tierney, D. and Bracken, J.J. (1993) Survival of salmon, *Salmo salar L.*, eggs planted in upland streams. *Aquaculture and Fisheries Management*, **24**, 791–6.

Kennedy, G.J.A. and Strange, C.D. (1982). Efficiency of electric fishing for salmonids in relation to river width. *Fisheries Management,* **12**, 55–60.

Kenward, R.E. and Hodder, K.H. (1996). Ranges V. An analysis system for biological location data. Institute of Terrestrial Ecology, Wareham.

Kesminas, V. and Virbickas, T. (2000) Application of an adapted index of biotic integrity to rivers of Lithuania. *Hydrobiologia*, **422/423**, 257–70.

Kestemont, P., Didier, J., Dewandel, F. and Micha, J.C. (1998) Preliminary results on the adaptation of a biotic index of fish integrity to a European river basin (The Meuse, Belgium). *Verh. Internationale Verein. Limnologie*, **26**, 1243–4.

Kilgour, B.W. and Barton, D.R. (1999) Associations between stream fish and benthos across environmental gradients in southern Ontario, Canada. *Freshwater Biology*, **41**, 553–6.

Kondolf, G.M. and Wolman, M.G. (1993) The sizes of salmonid spawning gravels. *Water Resources Research,* **29(7)**, 2275–85.

Krueger, C.C. and Waters, T.F. (1983) Annual production of macroinvertebrates in three streams of different water quality. *Ecology*, **64(4)**, 840–50.

Kubecka, J., Duncan, A., Duncan, W., Sinclair, D. and Butterworth, A. (1994) Brown trout populations of three Scottish lochs estimated by horizontal sonar and multimesh gillnets. *Fisheries Research*, **20**, 29–48.

Lee, G., and Jones, A. (1991) Effects of Eutrophication on Fisheries. *Reviews in Aquatic Sciences,* **5(3–4)**, 287–305.

Lobon-Cervia, J. and Utrilla, C.G. (1993) A simple model to determine stream trout (*Salmo trutta* L.) densities based on one removal with electrofishing. *Fisheries Research*, **15,** 369–78.

Lobon-Cervia, J. and Fitzmaurice, P. (1988) Stock assessment, production rates and food consumption in four contrasting Irish populations of brown trout (*Salmo trutta* L.). *Polskie Archivum Hydrobiologie*, **35**, 3–4.

Lucas, M.C., Mercer, T., Batley, E., Frear, P.A., Peirson, G., Duncan, A. and Kubecka, J. (1998) Spatio–temporal variations in the distribution and abundance of fish in the Yorkshire Ouse system. *The Science of the Total Environment*, **210**, 437–55.

MacLennan, D.N. and Holliday, D.V. (1996) Fisheries and plankton acoustics: past, present, and future. *ICES Journal of Marine Science*, **53(2)**, 513–16.

Maitland, P.S. and Campbell, R.N. (1992) *Freshwater Fishes*. Harper Collins, UK. 368pp.

Mason, C.F. (1996) Biology of Freshwater Pollution. Longman group UK, pp. 356.

McCarthy, D. and Kennedy, M. (1965) The Biology of Irish Coarse Fishes: A Synopsis of Published Irish, British and Continental Data and of Current Irish Research on the Species found in Ireland. Inland Fisheries Trust. Unpublished internal document.

McCormick, F.H., Hughes, R.M., Kaufmann, P.R., Peck, D.V., Stoddard, J.L. and Herlihy, A.T. (2001) Development of an Index of Biotic Integrity for the mid *Atlantic Highlands region. Transactions of the American Fisheries Society*, **130**, 857–77.

McCune B. and Mefford M. J. (1997) Multivariate Analysis of Ecological Data. MJM Software, Gleneden Beach Oregon, USA.

McCune, B. and Mefford, M. J. (1999) Multivariate Analysis of Ecological Data. Version 4. MJM Software, Gleneden Beach Oregon, USA.

McDonnell, N. (2005) The Relationship between Water Quality and Fish Stocks in Irish Rivers. PhD thesis presented to the National University of Ireland.

McGarrigle, M.L. (1998) Impact of eutrophication on Irish river water quality, in Wilson, J.G. (ed.) *Eutrophication in Irish Waters*, Royal Irish Academy, Dublin, 82–90.

McGarrigle, M.L. (2001) Eutrophication of Irish waters – from science to legislation and management. Went Memorial Lecture. *Occasional Papers in Irish Science and Technology*, Royal Dublin Society, **24,** 1–17.

McGarrigle, M.L., Bowman, J.J., Clabby, K.J., Lucey, J., Cunningham, P., MacCarthaigh, M., Keegan, M., Cantrell, B., Lehane, M., Cleneghan, C. and Toner, P.F. (2002) *Water Quality in Ireland* 1998–2000, Environmental Protection Agency, Wexford, Ireland.

McGinnity, P., Gargan, P., Roche, W., Mills, P. and McGarrigle, M. (2003) Quantification of the freshwater salmon habitat asset in Ireland using data interpreted in a GIS platform. *Irish Fisheries Ecology and Management Series*: Number 3, Central Fisheries Board, Dublin, Ireland.

McGovern, P. and McCarthy, T.K. (1992) Local movements of freshwater eels (*Anguilla anguilla* L.) in western Ireland, in Priede, I.G. and Swift, S.M. (eds) *Wildlife Telemetry: Remote Monitoring and Tracking of Animals, Chichester*, Ellis Horwood, 319–27.

Mebane, C.A., Maret, T.R. and Hughes, R.M. (2003) An index of biological integrity (IBI) for Pacific Northwest Rivers. *Transactions of the American Fisheries Society*, **132**, 239–61.

Milner, N.J., Wyatt, R.J., Barnard, S. and Scott, M.D. (1995) Variance structuring in stream salmonid populations, effect of geographical scale and implications for habitat models. *Bulletin Francais de la peche et pisciculture*, **337**, 387–439.

Miltner, R.J. and Rankin, E.T. (1998) Primary nutrients and the biotic integrity of rivers and streams. *Freshwater Biology*, **40**, 145–58.

Mitro, M.G. and Zale, A.V. (2000). Predicting fish abundance using single-pass removal sampling. *Canadian Journal of Fisheries and Aquatic Sciences*, **57**, 951–61.

Moss D., Furse M. T., Wright J. F. and Armitage P. D. (1987) The prediction of the macro-invertebrate fauna of unpolluted running-water sites in Great Britain using environmental data. *Freshwater Biology,* **17**, 41–52.

Norris, R.H. (1996) Predicting water quality using reference conditions and associated communities, in Bailey, R.C., Norris, R.H. and Reynoldson (eds) *Study Design and Data Analysis in Benthic macroinvertebrate Assessments of Freshwater Ecosystems using a reference site approach*, Technical Information Workshop, North American Benthological Society, 44th Annual Meeting, Kalispell, Montana, 32–52.

Oberdorff, T., Pont, D., Hugueny, B. and Chessel, D. (2001) A probalistic model characterizing fish assemblages of French rivers: a framework for environmental assessment. *Freshwater Biology,* **46,** 399–415.

O'Connor, W.C.K. (1998) The effects of siltation on Atlantic salmon, *Salmo salar* L., embryos in the river Bush. *Fisheries Management and Ecology,* **5**, 393–401.

O'Grady, M.F. (1981) Some direct gillnet selectivity tests for brown trout populations. *Irish Fisheries Investigations, Series A (Freshwater)*, **22**, 9pp.

O'Grady, M.F., King, J.J. and Curtin, J. (1991) The effectiveness of the physical instream works programmes in enhancing salmonid stocks in a drained Irish lowland river, in Mills, D. (ed.), *Strategies for the Rehabilitation of Salmon Rivers,* Linnaean Society, London.

Ottaway, E.M., Carling, P.A., Clarke, A. and Reader, N.A,. (1981) Observations on the structure of brown trout, *Salmo trutta* Linnaeus, redds. *Journal of Fish Biology,* **19,** 593-607.

Ovidio, M. and Phillippart, J.C. (2000) Methodological bias in home range and mobility estimates when locating radio-tagged trout, *Salmo trutta*, at different time intervals. *Aquatic Living Resources,* **13**, 449–54.

Parr, L.B. and Mason, C.F. (2003) Long-term trends in water quality and their impact on macroinvertebrate assemblages in eutrophic lowland rivers. *Water Research*, **37**, 2969–79.

Parsons, M. and Norris, R.H. (1996) The effect of habitat-specific sampling on biological assessment of water quality using a predicitive model. *Freshwater Biology,* **36**, 419–34.

Penczak, T. (1985) Influence of site area on the estimation of the density of fish populations in a small river. *Aquaculture and Fisheries Mangement*, **1,** 273–85.

Plafkin, J.L., Barbour, M.T. Porter, K.D. Gross, S. K. and Hughes, R.M. (1989). *Rapid Bioassessment Protocols for Use in Streams and Rivers: Benthic Macroinvertebrates and Fish*. EPA/444/4-89-001. US Environmental Protection Agency, Washington DC.

Pont, D., Hugueny, B., Beier, U., Goffaux, D., Melcher, A., Noble, R., Rogers, C., Roset, N. and Schmutz, S. (2006) Assessing river biotic condition at a continental scale: A European approach using functional metrics and fish assemblages. *Journal of Applied Ecology*, **43**, 70–80.

Reynolds, L., Herlihy, A.T., Kaufmann, P.R., Gregory, S.V. and Hughes, R.M. (2003) Electrofishing effort requirements for assessing species richness and biotic integrity in western Oregon streams. *North American Journal of Fisheries Management*, **23**, 450–61.

SAS (2000) *SAS/STAT Version 8.2*. Cary, North Carolina: SAS Institute Inc.

Schmutz, S., Kaufmann, M., Vogel, B., Jungwirth, M. And Muhar, S. (2000) A multi-level concept for fish-based, river-type-specific assessment of ecological integrity. *Hydrobiologia,* **422/423,** 279–89.

Scott, M.C. and Hall, Jr. L.W. (1997) Fish assemblages as indicators of environmental degradation in Maryland coastal plain streams. *Transactions of the American Fisheries Society*, **126(3)**, 349–60.

Simmonds, E.J. and MacLennan, D.N. (eds) (1996) Fisheries and Plankton Acoustics. Proceedings of an ICES International Symposium held in Aberdeen, Scotland 12–16 June 1996. *ICES Journal of Marine Science,* **53**, 535pp.

Simpson, J. and Norris, R. H. (2000) Biological assessment of water quality: development of AUSRIVAS models and outputs, in Wright, J. F., Sutcliffe, D.W. and Furse, M.T. (eds) *Assessing the Biological Quality of Freshwaters. RIVPACS and Other Techniques,* Ambleside, UK, Freshwater Biological Association, 125–42.

Smith, M.J., Kay, W.R., Edward D.H.D., Papas, P.J., Richardson, K.S.J., Simpson, J.C., Pinder, A.M., Cale, D.J., Horwitz, P.H.J., Davis, J.A., Yung, F.H., Norris, R.H. and Halse, S.A. (1999) AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. *Freshwater Biology,* **41**, 269–82.

Soulsby, C., Youngson, A.F., Moir, H.J. and Malcolm, I.A. (2001) Fine sediment influence on salmonid spawning habitat in a lowland agricultural stream: a preliminary assessment. *The Science of the Total Environment,* **265,** 295–307.

Southwood, T.R.E. and Henderson, P.A. (2000). *Ecological Methods,* Blackwell Science Publications, Oxford,

Strahler, A.N. (1952) Hyposmetric (area-altitude) analysis of erosional topography. *Bulletin of the Geological Society of America*, **63,** 1117–42.

ter Braak, C.J.F. and Prentice, I.C. (1988) A theory of gradient analysis. *Advances in Ecological Research,* **18,** 93–138.

Townsend, C.R., Hildrew, A.G. and Francis, J. (1983) Community structure in some southern English streams: the influence of physicochemical factors. *Freshwater Biology,* **13,** 521–44.

Turnpenny, A.W.H. and Williams, R. (1980) Effects of sedimentation on the gravels of an industrial river system. *Journal of Fish Biology*, **17**, 681–93.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. and Cushing, C.E. (1980) The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences,* **37(1)**, 130–7.

Walley, W.J. and Fontama, V.N. (2000) New approaches to river quality classification based upon Artificial intelligence, in Wright, J.F., Sutcliffe, D.W. and Furse, M.T. (eds) *Assessing the Biological Quality of Freshwaters. Rivpacs and other Techniques.* Freshwater Biological Association, UK.

Wallin, M., Wiederholm, T. and Johnson, R.K. (2003) *Guidance on Establishing Reference Conditions and Ecological Status Class Boundaries for Inland Surface Waters.* CIS Working Group 2.3–REFCOND.

Weatherley, N.S., Rogers, A.P., Goenaga, X. and Ormerod, S.J. (1990) The survival of early life stages of brown trout (Salmo trutta L.) in relation to aluminium speciation in upland Welsh streams. *Aquatic Toxicology,* **17,** 213–30.

Welcomme, R.L. (1985) *River Fisheries.* FAO Fisheries Technical Paper, **262,** 330pp.

Went, A.E.J. and Kennedy, M. (1976) *List of Irish Fishes.* Stationery Office, Dublin.

Williams, R.J., White, C., Harrow, M.L. and Neal, C. (2000) Temporal and small-scale spatial variations of dissolved oxygen in the river Thames, Pang and Kennet, UK. *The Science of the Total Environment,* **251/252**, 497–510.

Wooton, R.J. (1992) *Fish Ecology,* Blackie and Son Ltd, Glasgow.

Wright, J.F. (1995) Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. *Australian Journal of Ecology,* **20,** 181–97.

Wright, J.F., Moss, D., Armitage, P.D. and Furse, M.T. (1984) A preliminary classification of running water-sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. *Freshwater Biology,* **14**, 221–56.

Wright, J.F., Furse, M.T. and Armitage, P.D. (1993) RIVIPACS- a technique for evaluating the biological quality of rivers in the UK. *European Water Pollution Control,* **3,** 15–25.

Wyatt, R.J. (2002). Estimating riverine fish population size from single- and multiple-pass removal sampling using a hierarchical model. *Canadian Journal of Fisheries and Aquatic Sciences*, **59**, 695–706.

Young, M.K. (1999) Summer diel activity and movement of adult brown trout in high-elevation streams in Wyoming, USA. *Journal of Fish Biology*, **54**, 181–9.

Zalewski, M. and Cowx, I.G. (1990) Factors affecting the efficiency of electric fishing. In Cowx, I.G. and Lamarque, P. (eds), F*ishing with Electricity,* Fishing News Books, Blackwell Scientific Publications, Oxford.

Zar, J.H. (1999) *Biostatistical Analysis,* 4th edn, New Jersey, Prentice Hall.

Zippin, C. (1956). An evaluation of the removal method of estimating animal populations. *Biometrics,* **12,** 163–9.

Acronyms

AI	artificial intelligence
ANOSIM	analysis of similarity
ANOVA	analysis of variance
AUSRIVAS	Australian River Assessment System
CCA	canonical correspondence analysis
CEC	The European Commission
CEN	Comité Europeén de Normalisation
CFB	Central Fisheries Board
DCA	detrended correspondence analysis
DO	dissolved oxygen
ECOSTAT	ecogical status
EFI	European Fish Index
EPA	Environmental Protection Agency
FAME	Fish Based Assessment Method
GIS	geographical information system
HABSCORE	rapid habitat assessment method that uses a scoring system to rate habitat
IBI	Index of Biotic Integrity
ICI	Invertebrate Community Index
IFT	Inland Fisheries Trust
LSD	least significant difference
MRP	molybdate reactive phosphorus
MRPP	Multi-Response Permutation Procedure
O/E	Observed/Expected condition
RBD	River Basin District
RFB	Regional Fisheries Board
RIVPACS	River Invertebrate Prediction and Classification Scheme
TVG	time varied gain
TWINSPAN	two-way indicator species analysis
WFD	Water Framework Directive

Appendix 1

List of sites (including CFB archive sites) included in the EPA Q-values and fish project database. XY coordinates of each site are also included

Site code	Easting	Northing	Hydrometric area	Catchment	Subcatchment	River name	Site	Year of survey
CFB001	143115	278984	30	Mask	Robe	Robe	“Main channel, Bekan (u/s site)”	2001
CFB002	143082	278900	30	Mask	Robe	Robe	“Mai channel(, d/s site)”	2001
CFB003	141515	274906	30	Mask	Robe	Robe	“Main channel, u/s Brickens bridge”	2001
CFB004	141774	274675	30	Mask	Robe	Robe	“Main channel, d/s Brickens bridge”	2001
CFB005	130470	273075	30	Mask	Robe	Vincent Walsh’s	Mill site	2001
CFB006	130753	272563	30	Mask	Robe	Vincent Walsh’s	Campbells land	2001
CFB007	136549	269240	30	Mask	Robe	Ballindine	d/s town	2001
CFB008	132118	274669	30	Mask	Robe	Mayfield	u/s site	2001
CFB009	133509	273039	30	Mask	Robe	Mayfield	main road site (d/s br)	2001
CFB010	134930	272824	30	Mask	Robe	Claremorris golf	d/s 1st footbridge	2001
CFB011	138186	274783	30	Mask	Robe	Ballygowan	main road site	2001
CFB012	137283	275979	30	Mask	Robe	Ballygowan	Maggies house	2001
CFB013	141312	272537	30	Mask	Robe	Kilknock	d/s roadbridge	2001
CFB014	142240	275059	30	Mask	Robe	Brickens stream	opposite pub (d/s br)	2001
CFB015	142330	275348	30	Mask	Robe	Brickens stream	d/s cattle drink (d/s landbr)	2001
CFB016	138876	278605	30	Mask	Robe	Top tributaru	u/s bridge	2001
CFB017	180237	412589	38	Rosses	Dungloe	Dungloe stream	Bet. Craghy & upr. Craghy (u/s road)	2001
CFB018	180974	411914	38	Rosses	Dungloe	Dungloe stream 2	Site 2 Croveigh R. (u/s fish pass/waterfall)	2001
CFB019	180502	412423	38	Rosses	Dungloe	Dungloe stream	Top sxn of Craghy (across lake)	2001
CFB020	180332	412301	38	Rosses	Dungloe	Dungloe stream	R. Croveigh (u/s of lake)	2001
CFB021	176073	413751	38	Rosses	Meala	Meala stream 1	R. Meala (u/s of road bridge)	2001
CFB022	185065	409872	38	Rosses	Owennamarve	Owennamarve main	Adj. to road d/s of island & high gradient braided section	2001
CFB023	185032	409872	38	Rosses	Owennamarve	Owennamarve main	U/s of braided sxn and broken br (2nd br.) see photos	2001
CFB024	185674	410671	38	Rosses	Owennamarve	Owennamarve main	“D/s of landbridge on 2nd track, btm of site on riffle,”	2001
CFB025	184462	410627	38	Rosses	Dungloe	Owennamarve main	U/s road on bend of river down to bridge	2001
CFB026	183546	410030	38	Rosses	Dungloe	Dungloe stream	“Top site bside telegraph pole, btm approx 5m u/s of bridge”	2001
CFB027	181614	409032	38	Rosses	Dungloe	L. Sallagh trib.	“Btm net bside fence post LHS, u/s roadbridge u/s L. Sallagh”	2001
CFB028	183554	408029	38	Rosses	Owennamarve	Owennamarve main	“U/s roadbr., btm site at riffle u/s bge, top around bend”	2001
CFB029	180224	407366	38	Rosses	Owennamarve	Owennamarve main	“U/s roadbr., top site on top end of deep pool”	2001
CFB030	178898	412045	38	Rosses	Dungloe	Dungloe stream	D/s sluice bet. Craghy & Dunloe lake	2001
CFB031	178046	412940	38	Rosses	Dungloe	Dungloe stream	U/s side of roadbr. = bottom net	2001
CFB032	173426	413713	38	Rosses	Meala	Meala stream 2	U/s of road bridge	2001
CFB033	194899	349295	35	Melvin	County	County main chan	U/s foot bridge	2001
CFB034	196825	346933	35	Melvin	County	County main chan	“Top at 1st weir, bottom at 2nd weir, u/s 2 waterfalls”	2001

Site code	Easting	Northing	Hydrometric area	Catchment	Subcatchment	River name	Site	Year of survey
CFB035	197297	346060	35	Melvin	County	County main chan	D/s br. in Kittyclogher beside new houses	2001
CFB036	197879	346060	35	Melvin	County	County main chan	D/s roadbr. near Kittyclogher	2001
CFB037	194348	349998	35	Melvin	County	County main chan	Stopnet riffle immed. d/s staff gauge bside house down hill	2001
CFB038	196919	346752	35	Melvin	County	County trib	"1mm u/s waterfall nr. garda checkpoint, btm@ u/s end waterfa"	2001
CFB041	197319	352816	35	Melvin	Roogagh	Roogagh main cha	"D/s br. (immed. u/s net at u/s end br., under br.)"	2001
CFB042	89356	123128	23	Feale		Shanow	"D/s br. Kilflyn no top net, used weir at pool"	2001
CFB043	95331	119327	23	Feale		Glashareagh	"Trib. of Smearlagh, top net at riffle, d/s of br."	2001
CFB044	101311	126623	23	Feale		Smearlagh	20m d/s of Forans br.	2001
CFB045	107181	119524	23	Feale		Owveg	D/s of bridge 20m	2001
CFB046	107100	120294	23	Feale		Owveg	"Sectioned off stretch on RHS, gravel removal site"	2001
CFB047	107568	121730	23	Feale		Owveg	Site 6 under cliff	2001
CFB048	107803	122223	23	Feale		Owveg	10m d/s of Pitch br.	2001
CFB049	113192	118529	23	Feale		Clydagh	U/s bridge at Brosna village	2001
CFB050	117203	115248	23	Feale		Breanagh	Ken 061 341637	2001
CFB051	117433	119644	23	Feale		Caher	"Top net approx 30m d/s br. @ riffle, btm net @ nxt riffle"	2001
CFB052	116768	126850	23	Feale		Allaghaun	"D/s roadbr., above pumphouse, top net = 5m d/s of island"	2001
CFB053	115019	130925	23	Feale		Oolagh	"Immed. d/s br., btm net @ d/s end of riffle, d/s sycamore"	2001
CFB054	112578	135115	23	Feale		Galey	"D/s br. in Athea, btm net just above bend at end of riffle"	2001
CFB055	121707	115758	23	Feale		Feale main chann	Alongside road and footpath	2001
CFB056	118333	115129	23	Feale		Feale main chann	Photo looking u/s	2001
CFB057	113555	120277	23	Feale		Feale main chann	"U/s of confluence with Clydagh, gravel removal site"	2001
CFB058	110939	121651	23	Feale		Feale main chann	50m d/s of Glasnocoo confluence	2001
CFB059	110628	122140	23	Feale		Feale main chann	"D/s of trib on RHS.Main flow on LHS, large gravel divide"	2001
CFB060	107087	130695	23	Feale		Feale main chann	"Site RHS in main flow,btm 10m d/s weir(RHS),top= dglweir u/s"	2001
CFB061	104351	138401	23	Feale		Galey	D/s roadbr. top=top riffle d/s br. 1st br. on Listowel rd.	2001
CFB062	106863	137076	23	Feale		Galey	D/s rd br. & wier/fish pass near graveyard past Moyvare	2001
CFB063	254540	278185	26	Sheelin		Millbrook	U/s of landbr. behind Texaco garage	2001
CFB064	256628	279474	26	Sheelin		Summerbank	"Top of site = new cattle drink, bottom= marked by stake"	2001
CFB065	249559	280129	26	Sheelin		Halfcarton	"U/s of roadbr., btm=10m d/s weir & top=beside stile"	2001
CFB066	253268	288355	26	Sheelin		Pound	"From cattledrink/stake, d/s of br. opp. insurance place"	2001
CFB067	251051	291307	26	Sheelin		Mt. Nugent	"Mt.Nugent-Rassan R., D/s of road bridge"	2001
CFB068	288470	265651	7	Boyne	Boyne	Boyne main chann	d/s kilcairn bridge	2001
CFB069	287367	267509	7	Boyne	Boyne	Boyne main chann	u/s railway bridge	2001
CFB075	88009	89292	22	Laune		Dunloe	"Site 3,"	2001
CFB076	84669	88464	22	Laune		Owenacullin	"Site 6 just below 2 tribs join, d/s rd br."	2001
CFB078	84510	91240	22	Laune		Site 9	Site 9	2001

Site code	Easting	Northing	Hydrometric area	Catchment	Subcatchment	River name	Site	Year of survey
CFB079	84302	90405	22	Laune		Owenacullin	“Site 10,”	2001
CFB080	82521	88172	22	Laune		Glasheenrogoad	D/s greenbr.=150m gap ditch from pinkhouse like dugout drain	2001
CFB081	82851	88433	22	Laune		Gaddagh	“Site 12, upper reaches”	2001
CFB082	84165	91142	22	Laune		Gaddagh	“Site 16b,”	2001
CFB083	83960	90967	22	Laune		Gaddagh	“Site 16a, u/s site 16b”	2001
CFB084	80845	93431	22	Laune		Tullynascally	“Site 20,”	2001
CFB085	79986	90037	22	Laune		Meanus	Site 22	2001
CFB086	81077	91481	22	Laune		Meanus	“Site 23, u/s of ford”	2001
CFB087	80525	90082	22	Laune		Meanus	“Site 26, IV 80525, some rip rep on RHS”	2001
CFB088	80433	93822	22	Laune		Meanus	“Site 27 access via track RHS, u/s confluence”	2001
CFB089	76842	89029	22	Laune		Cottoners	“Site 29/Glencuttaun,”	2001
CFB090	76039	89555	22	Laune		Owennalacken	Site 30	2001
CFB091	77026	89810	22	Laune		Cottoners	“Site 31, u/s & d/s ford, d/s br.”	2001
CFB092	78669	93936	22	Laune		Cottoners	“Site 34, d/s br.”	2001
CFB093	98488	96756	22	Laune		Gweestin	“Site 37, NE of Cleedagh,”	2001
CFB094	94861	98090	22	Laune		Gweestin	“Site 39(Kilkneedan), fallup often flood”	2001
CFB095	96016	97097	22	Laune		Glanooragh	“Site 41, u/s br. top of site just below where trib joins RHB”	2001
CFB096	94154	96710	22	Laune		Glanooragh	Site 42	2001
CFB097	90765	97687	22	Laune		Glanooragh	“Site 45 u/s br.,”	2001
CFB098	92422	98289	22	Laune		Gweestin	U/s Gweestin Br. site 46	2001
CFB099	85856	97095	22	Laune		Gweestin	Site 50a below Listry br. at farmhse and intro 2 fields	2001
CFB100	83506	95957	22	Laune		Kealbrogeen	“Site 52 u/s of br. thro field on LHB, farmhse on otherside”	2001
CFB101	85586	95185	22	Laune		Dorannagh	“Site 53,”	2001
CFB102	89931	93567	22	Laune		Douglasha	“Site 57 , LHB of br.”	2001
CFB103	78545	89662	22	Laune		Skregbeg	“Site 61,”	2001
CFB104	76808	86751	22	Laune		Cottoners	“Site 62, fished u/s / d/s of bridge”	2001
CFB105	77042	87202	22	Laune		Cottoners	“Site 63, generating station outfall”	2001
CFB106	79312	91108	22	Laune		Cottoners	“Site 64, Skregbeg trib.”	2001
CFB107	121010	293670	34	Moy		Castlebar	“Exp site 5, d/s bridge”	2001
CFB108	122060	293700	34	Moy		Manulla	“Exp site, stop nets are at riffle”	2001
CFB109	111450	319539	34	Moy		Brandra	“Exp site 3, alongside road up to Marny Ben”	2001
CFB110	131400	287505	34	Moy		Pollagh	D/s br. in a riffle	2001
CFB111	135245	271474	30	Mask	Robe	Robe	“Main channel, u/s Five Arch bridge”	2001
CFB112	141402	272442	30	Mask	Robe	Kilknock	u/s road bridge	2001
CFB113	133384	267960	30	Mask	Robe	Scardaun	u/s road bridge	2001
CFB114	157636	264586	30	Corrib	Clare	Boyounagh	“D/s Boyounagh br. bet. 2 cattledrinks,Garveys Lynchs at farm”	1998
CFB115	157052	256216	30	Corrib	Grange	Ballydesmond	“Control, about 150 u/s of road br., u/s & adj to exp site”	1998
CFB116	157107	256232	30	Corrib	Grange	Ballydesmond	“Exp., riprap on both banks, nvr fenced still being trampled”	1998
CFB117	155986	254216	30	Corrib	Grange	Mahanagh	Control above Mahanagh br. longer than last yr now staked	1998
CFB118	155961	254157	30	Corrib	Grange	Mahanagh	Exp. below Mahanagh br. fort brown br. stone deflectors	1998
CFB119	151279	250432	30	Corrib	Grange	Omaun	Exp site 5 paired timber deflectors some ineffective	1998
CFB120	151092	250478	30	Corrib	Grange	Omaun	“Control site, about 80m u/s road br., see red stakes”	1998

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CFB121	153660	267639	30	Corrib	Clare	Flaska	Exp site 1 post works needs to be fenced to stop trampling	1998
CFB122	153735	267678	30	Corrib	Clare	Flaska	Control 1 above pumphouse bridge	1998
CFB123	152155	271410	30	Corrib	Clare	Cloonfad	Exp 2 bet 2 landbr. 2 stretches 1=36m frm d/s br. u/s to fen	1998
CFB124	152213	271419	30	Corrib	Clare	Cloonfad	"10m d/s from previous site, 2nd of 2 expiramental 1"	1998
CFB125	152806	271738	30	Corrib	Clare	Cloonfad	Exp1 sxn1 uppermost of 2 sxns	1998
CFB126	152468	271526	30	Corrib	Clare	Cloonfad	Control d/s from small br. with concrete plinth	1998
CFB127	140576	259386	30	Corrib	Clare	Knocknagar	Control 200m d/s br. btm=15m u/s confluence top=100m u/s	1998
CFB128	140968	258802	30	Corrib	Clare	Knocknagar	Ctrl btm=70m u/s br. top=20m u/s hawthorn on RHB	1998
CFB129	140630	258973	30	Corrib	Clare	Knocknagar	Control just u/s from br.	1998
CFB130	140690	259501	30	Corrib	Clare	Knocknagar	"1st 2 exp, btm net=15m above confluence with larger stream"	1998
CFB131	140923	259493	30	Corrib	Clare	Knocknagar	"Control 250m br. u/s to fence, same depth & width throughout"	1998
CFB132	109260	267834	30	Mask	Glensaul	Glensaul	"Site 9, top net=20m d/s from confluence with waterfall"	1998
CFB133	108014	267270	30	Mask	Glensaul	Glensaul	"Site 21, 60m u/s from ford"	1998
CFB134	106654	266434	30	Mask	Glensaul	Glensaul	"Zone 31, d/s of confluence Darranaderra trib."	1998
CFB135	106328	266265	30	Mask	Glensaul	Glensaul	Zone 32	1998
CFB136	105473	266167	30	Mask	Glensaul	Glensaul	"Zone 38, past house along divised path (beerbottles dumped)"	1998
CFB137	104690	265618	30	Mask	Glensaul	Glensaul	"Site 45, 20m d/s from uppermost br."	1998
CFB138	104356	265426	30	Mask	Glensaul	Glensaul	"Site 47, 50m d/s from uppermost br."	1998
CFB139	104231	265306	30	Mask	Glensaul	Glensaul	"Site 57, 50m above uppermost br."	1998
CFB140	106581	266496	30	Mask	Glensaul	Garranderra	Zone 1	1998
CFB141	106380	266649	30	Mask	Glensaul	Garranderra	"Zone 5, about 15m u/s from roadbr. to field"	1998
CFB142	106874	266740	30	Mask	Glensaul	Derryneeny		1998
CFB143	107945	267798	30	Mask	Glensaul	Tourmakeady moun	Mountain stream	1998
CFB144	104545	265650	30	Mask	Glensaul	Tourmakeady	"Zone 1, just below large waterfall"	1998
CFB144	109064	268044	30	Mask	Glensaul	Tourmakeady stre	Tourmakeady stream	1998
CFB145	108952	268547	30	Mask	Glensaul	Tourmakeady stre	"Zone 2, d/s of landbr. in forest 400m above large waterfall"	1998
CFB146	108537	268625	30	Mask	Glensaul	Tourmakeady stre	"Zone 3 chainage 20 top of channel, riffle d/s of high wtrfal"	1998
CFB147	104619	265595	30	Mask	Glensaul	Sruffaunagreeve	"Site 2, 30m u/s from confluence with river Glensaul"	1998
CFB148	104413	265839	30	Mask	Glensaul	Sruffaunagreeve	"Site 4, alongside riprap sxn on farm roadside"	1998
CFB149	104383	265917	30	Mask	Glensaul	Sruffaunagreeve	Site 5 @ tractor access beside house	1998
CFB150	104310	265991	30	Mask	Glensaul	Sruffaunagreeve	"Zone 6, alongside doused steel container behind barn"	1998
CFB151	104211	266094	30	Mask	Glensaul	Sruffaunagreeve	Site 7	1998
CFB152	104111	266163	30	Mask	Glensaul	Sruffaunagreeve	Site 8	1998
CFB153	111794	272469	30	Mask	Srah	Srah main channe	"Site 1, 20m d/s from disused br."	1998
CFB154	111672	272507	30	Mask	Srah	Srah	"Main Channel, Zone 5 site 2, 3m u/s from disused br."	1998
CFB155	111529	272906	30	Mask	Srah	Srah	"Main channel, Zone7site3 30m d/s from lwst br. before junc."	1998

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CFB156	111431	273114	30	Mask	Srah	Srah	Main channelSrah site 8 100m d/s of sharp bend	1998
CFB157	110914	272938	30	Mask	Srah	Srah	"Main channel, Upr Srah 10, u/s of br. at Sean Horan's house"	1998
CFB158	111400	273400	30	Mask	Srah	Derrinaderg	Zone 1 site 6 top net=5m d/s from Seanghort br.	1998
CFB159	111272	273850	30	Mask	Srah	Derrinaderg	Zone 2 site 7 btm net=weir u/s of br. to bend	1998
CFB160	111060	273830	30	Mask	Srah	Derrinaderg	Zone 3 site 8 (site 2)	1998
CFB161	110558	273785	30	Mask	Srah	Derrinaderg	Site 4 d/s of culvert br.	1998
CFB162	109546	273579	30	Mask	Srah	Derrinaderg	"Zone 4 site 10, u/s of top br."	1998
CFB163	111483	272550	30	Mask	Srah	Gortbunacullin	"Zone 1site 11, from foot bridge down 45m above Petes house"	1998
CFB164	111346	271907	30	Mask	Srah	Gortbunacullin	Site 2 below waterfall (zone 2 site 12)	1998
CFB165	109750	270917	30	Mask	Srah	Gortbunacullin	"Zone3 site 13, bottom net=1m above outflow"	1998
CFB166	112104	272988	30	Mask	Srah	Post office drai	Site 1 (zone 1 site 14)	1998
CFB167	112081	272451	30	Mask	Srah	Postoffice drain	"Zone 2 site 15, d/s 50m from rd"	1998
CFB168	114885	239556	30	Corrib	Drimneen	Drimneen	"Main channel, Zone 9 bottom net=20m u/s from footbridge belo"	1997
CFB169	114983	239414	30	Corrib	Drimneen	Drimneen	"Main channel, Zone10, top net=rocks 5m d/s landbr. btm=48m d"	1997
CFB170	115010	239289	30	Corrib	Drimneen	Drimneen	"Main channel, Site 11.1 u/s landbr. 300m in shaded area (ald"	1997
CFB171	114894	239119	30	Corrib	Drimneen	Drimneen	"Main channel, Zone 12"	1996
CFB172	114717	238964	30	Corrib	Drimneen	Drimneen	"Main channel, Just d/s br. on left adj. to Drimneen house d/"	1997
CFB173	114434	240598	30	Corrib	Drimneen	Magheramore	"Site 4, u/s of roadbridge d/s of railway bridge"	1997
CFB174	113549	241144	30	Corrib	Drimneen	Magheramore	Site 7 top=pole @ upr end forestry 2m below small drain/trib	1997
CFB175	92912	251615	30	Corrib	Failmore	Bunbhucain	Vortex weir-pre development experimental	1997
CFB176	93432	252222	30	Corrib	Failmore	Maum	Control 1yr post works at Mamean B&B	1998
CFB177	95422	251711	30	Corrib	Failmore	Failmore	Ctrl zone 4 opp. Mamean B&B braided channel wide floodplain	1998
CFB178	93048	251703	30	Corrib	Failmore	Bunbhucain	Exp post works xmas tree behind Mamean B&B @ western way	1998
CFB179	92904	251615	30	Corrib	Failmore	Bunbhucain	Control-vortex weir pre-development	1997
CFB180	122419	273331	30	Carra		Carrokilleen	U/s of Mullingar bridge	1998
CFB181	119183	279661	30	Carra		Annes stream	"Control, d/s of crossrd between 2 bridges (one unusual br.)"	1998
CFB182	118822	280462	30	Carra		Annes stream	Experimental	1998
CFB183	114873	279887	30	Carra		Ballintober	Zone 2 exp d/s bridge	1998
CFB184	115119	279330	30	Carra		Ballintober	Control 1st left after abbey u/s of bridge	1998
CFB185	118461	279227	30	Carra		Clogher	Exp d/s of br.into field(1st from lake)	1998
CFB186	253735	284812	26	Sheelin		Castlerahan (c)	U/s of rdbr. J kings site	1998
CFB187	250737	290795	26	Sheelin		Rassan	"Site 1 exp runs along by Mill house road, Crasserlough"	1998
CFB188	250712	290950	26	Sheelin		Rassan	Upr ctrl 500m d/s of woodland acros down lane & thro 2 field	1998
CFB189	250350	291150	26	Sheelin		Rassan	Upr exp in thro farmyard lane & 2 fields	1998
CFB190	255144	279172	26	Sheelin		Summerbank (E)	Summerbank exp site u/s of Millbrook rd br.	1998

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CFB191	256656	279493	26	Sheelin		Summerbank (C)	J. King site	1998
CFB192	251210	288380	26	Sheelin		Pound(Kildoragh)	See J. Kings sheets for details	1998
CFB193	253374	288996	26	Sheelin		Pound (KIlquilly	10m d/s of br. on Ballyjameduff rd. J. King site	1998
CFB194	252514	280936	26	Sheelin		Upper Inny	70m d/s of dam and fish pass	2001
CFB195	250083	288959	26	Sheelin		Finaway (E)	Exp prework 50m d/s from Mcguires piggery u/s 2 J. King site	1998
CFB196	246591	287586	26	Sheelin		Crover (C)	Control site 1 d/s main rd br.	1998
CFB197	247043	286757	26	Sheelin		Crover (E)	Site 2 exp green pogs	1998
CFB198	247101	286715	26	Sheelin		Crover (E)	Exp postworks gate opp Crover hse RHS field btm net=cattledk	1998
CFB199	241147	283843	26	Sheelin		Carrick (E)	Exp field with giant hrschestnut tree side rd off main rd	1998
CFB200	240972	285534	26	Sheelin		Maghera (C)	Down hill from chicken farm site2 contrl lwr grad. than sit1	1998
CFB201	241432	284303	26	Sheelin		Maghera (E)	Exp site 1 @ junction in rd near top of system	1998
CFB202	250610	289926	26	Sheelin		Mountnugent stre		2001
CFB202	250878	289286	26	Sheelin		Mountnugent (E)	Exp site Derrylea br. access thro farmyard on RHB	1998
CFB203	250786	289353	26	Sheelin		Mountnugent (C)	Control in through farmyard turn after M rd bridge	1998
CFB204	249915	291455	26	Sheelin		Rassan	"Rd from Greens mill past other sites, at T leftl of Rassan r"	1998
CFB205	99939	253911	30	Corrib	Cornamona	Cornamona	Ctrl 1 postworks rocks were arrangd to be on left bank	1998
CFB206	99801	253970	30	Corrib	Cornamona	Cornamona	"Exp1 20m u/s ctrl 1, 2 weirs constructed 1 fallen in (rubbl"	1998
CFB207	103395	253398	30	Corrib	Cornamona	Millpark	Immed below 2nd br u/s of Dooghta main chl up frm post offic	1998
CFB208	103412	252910	30	Corrib	Cornamona	Millpark	Site1 immed d/s 1st rd br u/s of dooghta main chl	1998
CFB209	103364	252963	30	Corrib	Cornamona	Millpark	Site 2 immed u/s of rd bridge	1998
CFB210	103263	253182	30	Corrib	Cornamona	Millpark	Site3	1998
CFB211	103398	253401	30	Corrib	Cornamona	Millpark	Site 4	1998
CFB212	103415	252913	30	Corrib	Cornamona	Millpark	Site 1 ctrl off rd immed d/s 1st rd br u/s of conflnce wt Co	1998
CFB213	103361	252970	30	Corrib	Cornamona	Millpark	Site 2 u/s of 1st rd br above confluence	1998
CFB214	103267	253186	30	Corrib	Cornamona	Millpark	Site 3	1998
CFB215	103398	253404	30	Corrib	Cornamona	Millpark	Site 4	1998
CFB216	100042	261585	30	Mask	Srahnalong	Srahnalong main	Site 1 d/s uppermost small br to the confluence	1998
CFB217	110398	268705	30	Mask	Treanlar	Treanlar	Site 1 d/s of main rd bridge	1998
CFB218	104248	257538	30	Mask	Cloughbrack	Cloughbrack main	70m d/s of br at America	1998
CFB219	103599	256101	30	Mask	Cloughbrack	Cloughbrack main	"Site5 riffle 100m d/s of bridge at Mill, site6 d/s pool"	1998
CFB220	103520	255995	30	Mask	Cloughbrack	Cloughbrack main	Site4 5m d/s of bridge at Mill	1998
CFB221	103438	255932	30	Mask	Cloughbrack	Cloughbrack main	Site3 u/s of bridge	1998
CFB222	103201	255838	30	Mask	Cloughbrack	Cloughbrack main	Site1 riffle Site2 d/s pool	1998
CFB223	103071	255820	30	Mask	Cloughbrack	Cloughbrack main	Site1 Petersburg hse between foot br. and road br.	1998
CFB224	108206	257036	30	Mask	Cahergall	Cahergall	Site2 150m d/s from main rdbr @ ford in field d/s oak on RHB	1998
CFB225	108042	256753	30	Mask	Cahergall	Cahergall	Site3 u/s of main rd bridge to weir	1998
CFB226	107989	256749	30	Mask	Cahergall	Cahergall	Site4 opp white bung u/s mainrd br top=wooden gate acros str	1998

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CFB227	110370	256838	30	Mask	Cloonbur	Cloonbur R.	Bottom forestry rd br before L Mask(d/s of this)	1998
CFB228	109691	256650	30	Mask	Cloonbur	Cloonbur R.	Top net at tree(willow LHB) 8m d/s of stone rd br in forestr	1998
CFB229	108462	255957	30	Mask	Cloonbur	Cloonbur R.	Maddog site 10m d/s landbr by cottage(1st left after Clonbur	1998
CFB230	107877	255632	30	Mask	Cloonbur	Cloonbur R.	Top=bend d/s 2nd br100m rd runs parallel btm=10m blw falls	1998
CFB231	107194	255394	30	Mask	Cloonbur	Cloonbur R. (	20m u/s 2nd br in forest(key from Barry Lamb Ballykine hse)	1998
CFB232	106919	255464	30	Mask	Cloonbur	Coolin L outflow	150m d/s lake btm=opp Rambler's walk stake 15m u/s dam	1998
CFB233	106824	255496	30	Mask	Cloonbur	Coolin L outflow	Lane thro clearfell forestry(lock)/ramblrs walk 50m d/s lake	1998
CFB234	106078	255188	30	Mask	Cloonbur	Coolin L trib	High rd turn right before hse site3 at oak & holly tree	1998
CFB235	105653	255512	30	Mask	Cloonbur	Coolin L trib	Kilbride d/s from 1st br from lake	1998
CFB236	135996	271661	30	Mask	Robe	Robe main channe	"Main channel, 400mu/s Castlemaggaretbr top=willow RHB"	1998
CFB237	141448	276245	30	Mask	Robe	Robe main channe	"Main channel, D/s 300m from Brickens rd br (C2 Brickens part"	1998
CFB238	141058	275264	30	Mask	Robe	Robe main channe	"Main channel, Postworks control u/s Brickens br u/s ford @ p"	1998
CFB239	141084	275168	30	Mask	Robe	Robe main channe	"Main channel, Exp D/s b/rock section (u/s Brickens)"	1998
CFB240	141414	274995	30	Mask	Robe	Robe main channe	"Main channel, Exp b/rock pools 98 post works 200m u/s Bricke"	1998
CFB241	132037	270306	30	Mask	Robe	Robe main channe	"Main channel, Control pre dev. period Curradoohey br(d/s Br)"	1998
CFB242	133730	270974	30	Mask	Robe	Robe main channe	"Main channel, D/s of Crossboyne bridge upper carb. limeston"	1998
CFB243	134976	271330	30	Mask	Robe	Robe main channe	"Main channel, Riprap on LHB & thalweg dug fished frm br u/s"	1998
CFB244	119083	263942	30	Mask	Robe	Robe main channe	"Main channel, just d/s weir d/s br at reddoor restaurant on"	1998
CFB245	74565	74591	21	Blackwater (K)		Blackwater (K)	"Main channel, Site1a exp post dev. just above road br."	1998
CFB246	74456	75233	21	Blackwater (K)		Blackwater (K)	"Main channel, Site1g @ ford at top of main channel (past sta"	1999
CFB247	81972	73317	21	Blackwater (K)		Dereendaragh	W. Roche's site 1yr post stepping of falls d/s site 21A	1999
CFB248	52230	63421	21	Currane	Finyglas	Finyglas	control site just u/s of landbr over Finyglas	1999
CFB249	51448	63910	21	Currane		Finglas	Exp site 1 d/s of ford u/s of br	1998
CFB250	56842	66364	21	Currane	Cloghvoola	Cloghvoola		1999
CFB250	52281	63360	21	Currane		Finglas		1998
CFB251	56837	66291	21	Currane	Cloghvoola	Cloghvoola	Site 4 between control and exp sites Cloghvoola Copall trib	1998
CFB252	58033	71421	21	Currane	Cummeragh	Cummeragh		1998
CFB253	57977	71336	21	Currane	Cummeragh	Cummeragh	Exp site Cummeragh 1 at 2nd ford across the river	1999
CFB254	58005	71392	21	Currane		Cummeragh	Cummeragh exp bend beside 2nd ford (at EU turn)	1998
CFB255	143115	278985	30	Mask	Robe	Robe	"Main channel, Site1 most u/s site weir in middle random boul"	2002
CFB256	143083	278901	30	Mask	Robe	Robe	"Main channel, Site2 d/s site btm=below weir below tree acros"	2002
CFB257	130468	273078	30	Mask	Robe	Vincent Walsh's	Mill site	2002

Site code	Easting	Northing	Hydrometric area	Catchment	Subcatchment	River name	Site	Year of survey
CFB258	130753	272565	30	Mask	Robe	Vincent Walsh's	"Vincent Walsh's bottom of site, across Campbells land"	2002
CFB259	136548	269244	30	Mask	Robe	Ballindine	d/s of landbridge and town	2002
CFB260	132119	274670	30	Mask	Robe	Mayfield	"u/s site, top of site at weir d/s br."	2002
CFB261	133511	273042	30	Mask	Robe	Mayfield	Beside Claremorris rd bet. 3 weirs	2002
CFB262	135113	270198	30	Mask	Robe	Golf club	Between i1st & 2nd footbr. d/s Clare lake	2002
CFB263	138184	274789	30	Mask	Robe	Ballygowan	main road site (Claremorris-Roscommon main road)	2002
CFB264	137283	275979	30	Mask	Robe	Ballygowan	Maggies House	2002
CFB265	141313	272539	30	Mask	Robe	Kilknock	d/s roadbridge	2002
CFB266	142241	275061	30	Mask	Robe	Brickens	opposite pub immed. u/s of bridge	2002
CFB267	142331	275350	30	Mask	Robe	Brickens	stopnet @cattledrink(btm) btm net@ deflector/weir(top	2002
CFB268	138875	278608	30	Mask	Robe	Top tributary	"u/s bridge, top of site is at br., bottom is at d/s end of c"	2002
CFB269	133385	267962	30	Mask	Robe	Scardaun	"u/s road bridge, top=riffle d/s cattledrk btm=riffle bside a"	2002
CFB270	141516	274911	30	Mask	Robe	Robe	"Main channel, u/s Brickens bridge (bottom=cattledrink)"	2002
CFB271	141773	274682	30	Mask	Robe	Robe	"Main channel, d/s Brickens br.bridge (u/s catttledrink) d/s"	2002
CFB272	135249	271470	30	Mask	Robe	Robe	"Main channel, d/s Castlemagaret landbridge, top= immediatel"	2002
CFB273	134982	271364	30	Mask	Robe	Robe	"Mian channel, u/s of Five Arch br."	2002
CFB274	134917	271259	30	Mask	Robe	Robe	"Main channel, riffle d/s of 1st weir d/s of Five Arches br."	2002
CFB275	261409	235020	7	Boyne	Boyne	Boyne	"Main channel, D/s Kinnegad br. 1m d/s of treatment works"	2002
CFB276	312719	196679	10	Avoca	Glendasan	Glendasan	"Beside Glendalough visitors centre, d/s bridge top net @ rif"	2002
CFB277	311121	197559	10	Avoca	Glendasan	Glendasan	"St. Kevins rd, on bend , top d/s 3 pine trees"	2002
CFB278	314392	196604	10	Avoca	Glenmacnass	Glenmacnass	"Behind Lynhams pub ,btm net is at top weir/falls"	2002
CFB279	305558	186149	10	Avoca	Ow	Ow trib	"Aghavanagh br., from bridge up to 3rd weir"	2002
CFB280	307297	193554	10	Avoca	Avonbeg	Avonbeg	"d/s landbridge, Top net @ d/s end br btm site @ riffle d/s t"	2002
CFB281	305597	185617	10	Avoca	Ow	Ow trib	"Ballygobban br. , top of site @ small weir d/s bridge"	2002
CFB282	312479	182168	10	Avoca	Ow	Ballycreen	"U/s Macreddin br., bottom of site is at bridge top is 30m u/"	2002
CFB283	74411	251218	32	Owenglin	Owenglin	Owenglin	"d/s granite/marble quarry, u/s of bridge, bottom of site is"	2002
CFB284	110099	281071	30	Mask	Aille	Camoge	d/s of landbridge -riffle on bend	2002
CFB285	112259	280124	30	Mask	Aille	Aille	d/s of bridge/ top net is at top of riffle	2002
CFB286	67464	250445	32	Owenglin	Owenglin	Owenglin	top is at weir d/s of main road bridge opposite B&B	2002
CFB287	138814	326520	35	Easky	Easky	Gowlan	"d/s collapsed bridge, bottom of site=u/s end of boulder @cen"	2002
CFB288	143748	332481	35	Dunneill	Dunneill	Dunneill	"u/s bridge, 2km u/s Dromore west, top of site is at weir u/"	2002
CFB289	109361	333274	33	Ballinglen	Ballinglen	Keerglen	"u/s track, Btm of site on riffle opp. track top=u/s of pool"	2002
CFB290	132533	317176	34	Moy	Srufaungal	Behy	"U/s br idge on lane ,bottom of site=gravel shoal immed u/s o"	2002
CFB291	148125	164158	27	Owenagarney	Owenagarney	Gourna	U/s of bridge (approx 8m)	2002
CFB292	138586	180876	27	Fergus	Fergus	Spancelhill	Bottom is at u/s end of bridge (small pool here)	2002

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CFB293	161027	172088	27		Glenomra	Broadford	Top of site is at d/s end of bridge above drain	2002
CFB294	183725	168341	25	Mulkear	Newport	Newport	"Top of site at weir u/s roadbridge, bottom is d/s end of rif"	2002
CFB295	316084	206206	10	Avoca	Cloghoge	Cloghoge	Top of site is at small weir u/s bridge	2002
CFB296	126455	310465	34	Moy	Carrowkerribly	Attymass	Site1 (chg. 1/50)-open site) u/s br. u/s cattledrk@ large a	2002
CFB297	127442	311263	34	Moy	Carrowkerribly	Attymass	Site 2 (chg. 17A-open site) Btm site=u/s bend bside tall ald	2002
CFB298	127447	311312	34	Moy	Carrowkerribly	Attymass	Site 3 (chg. 17B-closed) Begins at riffle u/s side chanl & t	2002
CFB299	127653	311675	34	Moy	Carrowkerribly	Attymass	Site 4 (chg. 25A-closed) d/s landbr top is break d/s br. & b	2002
CFB300	127639	311724	34	Moy	Carrowkerribly	Attymass	"Site 5 (chg. 25B-open)Top is u/s landbridge, bottom is riffl"	2002
CFB301	127624	311827	34	Moy	Carrowkerribly	Attymass	Site 6 (chg. 26/50-closed) D/s roadbr. begins from middle ca	2002
CFB302	127698	311989	34	Moy	Carrowkerribly	Attymass	Site 7 (chg. 27A-closed) Immed. u/s roadbr. and above cattle	2002
CFB303	127727	312018	34	Moy	Carrowkerribly	Attymass	Site 8 (chg. 27B-closed) Blockage at top of site begins @ ri	2002
CFB304	127830	312121	34	Moy	Carrowkerribly	Attymass	"Site 9 (chg. 31-shaded)U/s landbr, deep site with v-shaped c"	2002
CFB305	127933	312484	34	Moy	Carrowkerribly	Attymass	"Site 10 (chg. 34/50) Begins @ cattledrink on LHS, bottom is"	2002
CFB306	143115	278984	30	Mask	Robe	Robe main channe	"Main channel, Bekan (u/s site)"	2003
CFB307	143083	278901	30	Mask	Robe	Robe main channe	"Main channel, Bekan (d/s site)"	2003
CFB308	130471	273079	30	Mask	Robe	Vincent Walsh's	Mill site	2003
CFB309	130754	272567	30	Mask	Robe	Vincent Walsh's	Campbells land	2003
CFB310	136548	269242	30	Mask	Robe	Ballindine	d/s town	2003
CFB311	132119	274671	30	Mask	Robe	Mayfield	u/s site	2003
CFB312	133509	273038	30	Mask	Robe	Mayfield	main road site (d/s br)	2003
CFB313	134930	272824	30	Mask	Robe	Claremorris golf	d/s 1st footbridge	2003
CFB314	138189	274785	30	Mask	Robe	Ballygowan	main road site	2003
CFB315	137283	275979	30	Mask	Robe	Ballygowan	Maggies house	2003
CFB316	141314	272539	30	Mask	Robe	Kilnock	d/s roadbridge	2003
CFB317	142241	275062	30	Mask	Robe	Brickens stream	opposite pub (d/s br)	2003
CFB318	142331	275350	30	Mask	Robe	Brickens stream	d/s cattle drink (d/s landbr)	2003
CFB319	133386	267962	30	Mask	Robe	Scardaun	"u/s road bridge, top=riffle d/s cattledrk btm=riffle bside a"	2003
CFB320	214896	324812	36	Swanlinbar	Swanlinbar	Swanlinbar	Immediately d/s of tullydermot Falls	2003
CFB321	192896	418958	38	Devlin	Cronaniv Burn	Cronaniv Burn	Site is immediatelu u/s & d/s of Bridge u/s Dunlewy Lough	2003
CFB322	194850	413973	38		Gweebarra	Gweebarra	site is immediately u/s of Pollgrlass bridge	2003
CFB323	236947	446083	40	Clonmany	Ballyhallan	Ballyhallan	"top of site is d/s cattle drink, d/s road bridge"	2003
UCC001	183993	103802	18	Araglin		Douglas		2001
UCC002	184014	103937	18	Araglin		Douglas		2001
UCC003	184203	104100	18	Araglin		Douglas		2001
UCC004	184354	104254	18	Araglin		Douglas		2001
UCC005	184401	104753	18	Araglin		Douglas		2001
UCC006	184452	104791	18	Araglin		Douglas		2001
UCC007	184504	105004	18	Araglin		Douglas		2001
UCC008	184316	104562	18	Araglin		Douglas		2001
UCC009	184327	104675	18	Araglin		Douglas		2001
UCC010	184432	104776	18	Araglin		Douglas		2001
UCC011	184486	104843	18	Araglin		Douglas		2001

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UCC012	184492	104951	18	Araglin		Douglas		2001
UCC013	184538	105053	18	Araglin		Douglas		2001
UCC014	184565	105132	18	Araglin		Douglas		2001
UCC015	184601	105251	18	Araglin		Douglas		2001
UCC016	163350	71125	19	Lee		Curraheen		2001
UCC017	162912	71084	19	Lee		Curraheen		2001
UCC018	162783	70852	19	Lee		Curraheen		2001
UCC019	162846	70342	19	Lee		Curraheen		2001
UCC020	163256	69958	19	Lee		Curraheen		2001
UCC021	168803	69934	19	Lee		Curraheen		2001
UCC022	162292	67776	19	Lee		Curraheen		2001
UCC023	161956	69447	19	Lee		Curraheen		2001
UCC024	161649	69455	19	Lee		Curraheen		2001
UCC025	161449	69269	19	Lee		Curraheen		2001
UCC026	161199	69086	19	Lee		Curraheen		2001
UCC027	161011	68797	19	Lee		Curraheen		2001
UCC028	160943	68670	19	Lee		Curraheen		2001
UCC029	161006	68322	19	Lee		Curraheen		2001
UCC030	161257	68159	19	Lee		Curraheen		2001
UCC031	181699	153965	25	Mulkear	Bilboa	Glashacloonavere	Br. near Shanacloon	2002
UCC032	87755	58298	21	Glengariff	Glengariff	Glengariff	"Crossterry, below Caha pass"	2002
UCC033	180444	123442	16	Suir	Aherlow	Aherlow	d/s br. N of Anglesborough	2002
UCC034	130760	88273	18	Blackwater (Muns	Rathcool	Owenbaun	d/s Owenbaun br.	2002
UCC035	71190	86458	22	L. Caragh	Caragh	Caragh	Blackbones br.	2002
UCC036	117287	161980	27	Shannon estuary	Cloon	Cloon	Br. N of cranny	2002
UCC037	222984	104814	17	Sea at Dungarven	Colligen	Colligen	u/s of Scart bridge	2002
UCC038	68749	80650	22	L. Caragh	Coomnacarig	Coomnacarig	u/s of Coomnacarrig br.	2002
UCC039	133296	89497	18	Blackwater (Muns	Rathcoole	Ivale	Finnanfield br.	2002
UCC040	116111	75918	19	Lee	Sullane	Sullane	Milleeny br.	2002
UCC041	197064	119652	16	Suir	Tar	Glennyreea	2nd br. NE of Burncourt	2002
UCC042	104807	180195	28	Carrowkeel	Cloonbony	Cloonbony	Cloonbony br.	2002
UCC043	158770	46048	20	Ballinspittle	Ballinspittle	Ballinspittle	d/s br. in Ballinspittle town	2002
UCC044	143227	57894	20	Bandon	Sall	Ballymahan	alongside road nw Bandon	2002
UCC045	114699	104726	18	Blackwater	Blackwater	Blackwater	ford N Ballydesmond	2002
UCC046	160336	111746	18	Blackwater	Awbeg	Bregoge	Streamhill br. (N. Doneraile)	2002
UCC047	107151	104713	22	Maine	Brown Flesk	Brown Flesk	D/s br east of Scataglin	2002
UCC048	221955	136401	16	Suir	Anner	Clashawley	Saucestown br. NE Fethard	2002
UCC049	136047	42705	20	Fealge	Fealge	Fealge	Alongside road NW from Clonaki	2002
UCC050	138066	67229	19	Lee	Cummer	Cummer	Picnic area NW of Kilmurry	2002
UCC051	184022	103955	18	Araglin	Douglas	Douglas	Picnic area near army barracks	2002
UCC052	217031	101420	18	Blackwater	Finisk	Finisk	d/s br. near Millstreet	2002
UCC053	175698	87328	18	Blackwater	Bride	Flesk	"Br. at Condonstown, NW Watergr"	2002
UCC054	100213	44084	21	Fourmilewat er	Durrus	Durrus	u/s bridge on Sheephead walk	2002
UCC055	235066	132546	16	Suir	Lingaun	Lingaun	Whitehall Br (W. Grangemockler	2002
UCC056	175509	126419	24	Maigue	Morningstar	Morningstar	d/s br. NE Garryspillane	2002
UCC057	134618	108551	18	Blackwater	Dalua	Owenanare	Priory bridge	2002
UCC058	104375	56734	21	Owvane	Owenbeg	Owenbeg	u/s 2nd bridge	2002
UCC059	123506	39222	20	Roury	Roury	Roury	parallel to road	2002
UCC060	150176	82600	19	Lee	Shournagh	Shournagh	across from grotto	2002
UCC061	194027	79888	18	Blackwater	Womanagh	Kiltha	Br. NE Dungourney	2002
UCC062	152988	87892	18	Blackwater	Clyda	Ahadallane	alongside road leading N from	2002

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UCC063	200112	77350	18	Blackwater	Womanagh	Dissour	Picnic area NW Killeagh	2002
UCC064	190959	90899	18	Blackwater	Bride	Douglas	Athafola bridge	2002
UCC065	152997	46701	20	Kilbrittain	Kilbrittain	Kilbrittain	u/s picnic area	2002
UCC066	167518	124598	24	Maigue	Loobagh	Loobagh	alongside road NW of Kilfinnan	2002
UCC067	192155	129320	16	Suir	Aherlow	Rossadrehid	u/s Dromamarka br.	2002
UCC068	185636	143776	25	Mulkear	Dead	Cauteen	u/s Pope's Bridge	2002
UCC069	170649	136258	24	Maigue	Camoge	Mahore	d/s bridge N. of Hospital	2002
UCD001	282445	244846	9	Liffey	Rye Water	Ardrums Little	Rye Water trib.	2001
UCD002	282423	244964	9	Liffey	Rye Water	Ardrums little t	Rye Water trib	2001
UCD003	282416	244822	9	Liffey	Rye Water	Ardrums Great	Rye Water trib	2001
UCD004	285745	244000	9	Liffey	Rye Water	Gallow stream		2001
UCD005	285657	241905	9	Liffey	Rye Water	Rye Water	"Main channel, McLochlins Br."	2001
UCD006	288107	240591	9	Liffey	Rye Water	Rye Water	"Main channel, Balfeaghan Br."	2001
UCD007	293629	237710	9	Liffey	Rye Water	Lyreen	Main channelTown centre	2001
UCD008	293822	237681	9	Liffey	Rye Water	Lyreen	"Main channel, Seminary"	2001
UCD009	293242	237740	9	Liffey	Rye Water	Lyreen	"Main channel, College"	2001
UCD010	292986	239465	9	Liffey	Rye Water	Rye Water	"Main channel, Annes Br."	2001
UCD011	297968	237597	9	Liffey	Rye Water	Rye Water	"Main channel, S1E (d/s Sandfords br.)"	2001
UCD012	298011	237592	9	Liffey	Rye Water	Rye Water	"Main channel, S1C (d/s Sandfords Br."	2001
UCD013	292806	239872	9	Liffey	Rye Water	Moyglare	Rye Water trib.	2001
UCD014	294448	239229	9	Liffey	Rye Water	Moygaddy	Rye Water trib.	2001
UCD015	295966	239665	9	Liffey	Rye Water	Offaly bridge	Rye Water trib.	2001
UCD017	297510	237160	9	Liffey	Rye Water	Blakestown	Rye Water trib.	2001
UCD018	285712	235314	9	Liffey	Lyreen	Clonshanbo	Lyreen trib at Aghafullim	2001
UCD019	285090	233159	9	Liffey	Lyreen	Baltracey	"Lyreen trib, Telfers Br."	2001
UCD020	287220	233934	9	Liffey	Lyreen	Baltracey	"Lyreen trib, Baltracey br."	2001
UCD021	288999	234360	9	Liffey	Rye Water	Lyreen trib.	Frayne's Br.	2001
UCD022	289352	232581	9	Liffey	Rye Water	Lyreen	"Main channel, Connellys Br."	2001
UCD023	291660	219749	9	Liffey		Morell stream	egg expt site	2001
UCD024	292256	227239	9	Liffey		Morell stream	abattoir	2001
UCD025	293991	224356	9	Liffey		Painestown	near bridge	2001
UCD026	294167	224441	9	Liffey		Painestown	across field	2001
UCD027	292368	226444	9	Liffey		Painestown	Painestown near Morell	2001
UCD028	288584	228608	9	Liffey		Gollymochy strea	Clane	2001
UCD030	285640	219982	9	Liffey		Canal supply	Halverstown	2001
UCD031	284703	209179	9	Liffey		Kilcullen	Cemetary	2001
UCD032	291205	208917	9	Liffey		Lemonstown	Near house (Ardinode)	2001
UCD033	291214	208850	9	Liffey		Lemonstown	near rod.	2001
UCD034	292747	209757	9	Liffey	Liffey	Liffey main chan	Ballymore br.	2001
UCD035	302779	230988	9	Liffey		Griffin stream		2001
UCD034	319496	155571	11	Owenavorragh	Owenavorragh	Aughboy	Courtown	2002
UCD035	307267	251756	8	Broadmeadow	Broadmeadow	Broadmeadow	d/s Milltown bridge	2002
UCD036	307432	231767	9	Camac	Camac	Camac	d/s bridge	2002
UCD037	319148	240053	9	Santry	Santry	Santry	u/s shopping centre	2002
UCD038	270041	230477	14	Barrow	Figile	Ballyshannow	u/s Ticknevin br.	2002
UCD039	299038	237914	9	Liffey	Rye Water	Kellystown	"Rye water trib, d/s road bridge"	2002
UCD040	285712	235315	9	Liffey	Lyreen	Clonshambo	"Lyreen trib., Aghafullim (d/s of bridge)"	2002
UCD041	322215	241429	9	Mayne	Mayne	Mayne	Hole in the wall br.	2002
UCD042	301687	241726	9	Tolka	Tolka	Tolka	d/s Rusk br.	2002
UCD043	286347	201580	14	Barrow	Greese	Greese	n/w of Crosskeys	2002
UCD044	293709	237780	9	Liffey	Rye Water	Lyreen	"Main channel, Maynooth town centre"	2002
UCD045	304998	242636	9	Tolka	Pinkeen	Pinkeen	Br. S. of Calliagawee	2002
UCD046	310098	245703	8	Broadmeadow	Ward	Ward	d/s Coolatruth bridge	2002
UCD047	298011	237595	9	Liffey	Rye Water	Rye Water	"Main channel, d/s Sandfords bridge"	2002

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UCD048	289001	234365	9	Liffey	Rye Water	Lyreen trib.	"Lyreen trib., d/s Frayne's bridge"	2002
UCD049	291660	219751	9	Liffey	Morell	Morell	d/s Morell bridge	2002
UCD050	293991	224367	9	Liffey	Painestown	Painestown	at bridge	2002
UCD051	285642	219991	9	Liffey	Canal supply	Canal supply	Halverstown Xroads	2002
UCD052	288114	240599	9	Liffey	Rye Water	Rye Water	"Main channel, d/s Balfeaghan bridge"	2002
UCD053	294169	224459	9	Liffey	Painestown	Painestown	across field	2002
UCD054	305505	221227	9	Liffey	Brittas	Brittas	game reserve	2002
UCD055	315881	269388	7	Boyne	Nanny	Mosney	d/s road bridge	2002
UCD056	320306	214853	10	Dargle	Glencree	Glencree	u/s Onagh bridge	2002
UCD057	293915	221684	9	Liffey	Hartwell	Hartwell	br. NE Johnstown (Rathmore)	2002
UCD058	329117	204028	10	Newcastle	Newcastle	Newcastle	0.5km d/s Newcastle br.	2002
UCD059	308966	224407	9	Liffey	Dodder	Dodder	main road (water treatment pla	2002
UCD060	316952	265810	7	Boyne	Delvin	Delvin	br. near bridgefort	2002
UCD061	307536	175009	10	Avoca	Derrywater	Derrywater	"Drummin, u/s bridge"	2002
UCD062	284713	209182	9	Liffey	Kilcullen	Kilcullen	near graveyard	2002
UCD063	291219	208865	9	Liffey	Lemonstown	Lemonstown	Ardinode east	2002
UCD064	313313	223898	9	Liffey	Dodder	Owendoher	u/s Rockbrook	2002
UCD065	309478	223453	9	Liffey	Dodder	Dodder trib	u/s Piperstown	2002
UCD066	320726	213429	10	Dargle	Dargle	Dargle	Br. S. of Ballinagee	2002
UCD067	324927	207791	10	Newtownmou ntkenn	Newtownmount kenn	Newtownmountk enn	br. near ballyhorsey	2002
UCD068	322126	210463	10	Dargle	Vartry	Vartry trib	u/s Ballinasloe br.	2002
UCD069	299839	183640	12	Slaney	Derreen	Derreen	Ballykilmurray	2002
UCD070	319323	203950	10	Dargle	Vartry	Vartry trib	n. of Roundwood	2002
UCD071	298681	194892	12	Slaney	Little Slaney	Knickeen	Just inside Military range d/s Knickeen ford	2002
UCD072	294874	192367	12	Slaney	Little Slaney	Little Slaney	D/s Rosstyduff br. opp. army barracks on bend	2002
UCD073	297613	192443	12	Slaney	Little Slaney	Little Slaney	"d/s ford nw of Coan, through field on left (park before ford"	2002
UCD074	319264	219349	10	Dargle	Glencullen	Glencullen	d/s Glencullen br.	2002
UCD075	302450	215917	9	Liffey	Ballyward	Ballyward	beside Ballyward br.	2002
UCD076	286392	148597	12	Slaney	Urrin	Urrin	u/s of Ballycrystal bridge	2002
UCD077	289762	143729	12	Slaney	Urrin	Urrin trib	s. of curraghgraigue	2002
UCD078	297512	237193	9	Liffey	Rye Water	Balkestown	"Rye Water trib, Blakestown"	2002
UCD079	295968	239696	9	Liffey	Rye Water	Offaly bridge	"Rye Water trib, Offaly bridge"	2002
UCD080	294440	239229	9	Liffey	Rye Water	Moygaddy	Rye Water tribMoygaddy	2002
UCD081	292989	239472	9	Liffey	Rye Water	Rye Water	"Main channel, Annes bridge"	2002
UCD082	292808	239894	9	Liffey	Rye Water	Moyglare	"Rye Water trib, Moyglare"	2002
UCD083	282402	244823	9	Liffey	Rye Water	Ardrums Great	Ardrums Great	2002
UCD084	282446	244854	9	Liffey	Rye Water	Ardrums Little	Ardrums Little	2002
UCD085	282421	244968	9	Liffey	Rye Water	Ardrums little t	Ardrums little trib	2002
UCD086	287220	233940	9	Liffey	Lyreen	Baltracey	"Lyreen trib, d/s Baltracey bridge"	2002
UCD087	285090	233160	9	Liffey	Lyreen	Baltracey	"Lyreen trib, d/s telfers bridge"	2002
UCD088	302775	230994	9	Liffey	Griffin	Griffin		2002
UCD089	289358	232574	9	Liffey	Rye Water	Lyreen	Main channel (Connollys br.	2002
UCD090	292259	227251	9	Liffey	Morell	Morell	Morell br (old) near abbattoir	2002
UCD091	292367	226448	9	Liffey	Morell	Painestown	s. of Morell br. (old)	2002
UCD092	293624	237713	9	Liffey	Rye Water	Lyreen	"Main channel, Maynooth Seminary"	2002
UCD093	291214	208850	9	Liffey	Lemonstown Ardin	Lemonstown Ardin		0
UCD094	285745	244000	9	Liffey	Rye Water	Gallow		0
UCD095	285657	241905	9	Liffey	Rye Water	McLochlins		0
UCD096	297968	237597	9	Liffey	Rye Water	Rye Water	"Main Channel, d/s Sandfords bridge, S1E"	2002
UCG001	150381	219228	29	Dunkellin	Aggard	Aggard	d/s Aggard br.	2001
UCG002	150527	216822	29	Dunkellin	Aggard	Aggard Beg	near Holy well	2001

Site code	Easting	Northing	Hydrometric area	Catchment	Subcatchment	River name	Site	Year of survey
UCG003	151217	215939	29	Dunkellin	Aggard	Aggard	S. Ballylin west	2001
UCG004	154429	221241	29	Dunkellin	Craughwell	Craughwell trib	Boleydarragha	2001
UCG005	157916	219745	29	Dunkellin		St. Clerens	d/s Killilan br.	2001
UCG008	162657	222275	29	Dunkellin	Craughwell	Craughwell trib	d/s Turoe br.	2001
UCG009	169598	219635	29	Dunkellin	Craughwell	Craughwell trib	d/s Kilreekill br.	2001
UCG010	169730	220724	29	Dunkellin	Dunkellin	Craughwell trib	Carnaun br.	2001
UCG011	154627	223265	29	Dunkellin		Doovertha R.	d/s Rattys Br.	2001
UCG012	162673	226669	29	Dunkellin	Rafford	Cloghervaun R.	u/s Brackloon	2001
UCG013	165671	227467	29	Dunkellin	Rafford	Rafford	SE of Gortnaboha	2001
UCG014	165638	227148	29	Dunkellin	Rafford	Rafford	Knockboley	2001
UCG015	168553	226367	29	Dunkellin	Rafford	Cloghervaun R.	NE Brackloon	2001
UCG016	166297	230498	29	Dunkellin	Rafford	Rafford	S. of Coppanagh	2001
UCG017	168872	230745	29	Dunkellin	Rafford	Rafford	d/s Woodlawn br.	2001
UCG018	168674	231394	29	Dunkellin	Rafford	Rafford	Cloonahinch	2001
UCG019	150358	219218	29	Dunkellin	Aggard	Aggard	d/s Aggard br.	2002
UCG020	151218	215938	29	Dunkellin	Dunkellin	Aggard	S. Ballylin west	2002
UCG021	154413	221280	29	Dunkellin	Craughwell	Craughwell trib	Boleydarragha	2002
UCG022	157944	219742	29	Dunkellin		St. Clerens	d/s Killilan br.	2002
UCG023	162663	222259	29	Dunkellin	Craughwell	Craughwell trib	d/s Turoe br.	2002
UCG024	169584	219672	29	Dunkellin	Craughwell	Craughwell trib	d/s Kilreekill br.	2002
UCG025	169724	220756	29	Dunkellin	Dunkellin	Craughwell trib	Carnaun br.	2002
UCG026	154623	223272	29	Dunkellin		Doovertha R.	d/s Rattys Br.	2002
UCG027	162677	226668	29	Dunkellin	Rafford	Rafford	u/s Brackloon	2002
UCG028	165669	227502	29	Dunkellin	Rafford	Rafford	SE of Gortnaboha	2002
UCG029	165634	227188	29	Dunkellin	Rafford	Rafford	Knockboley	2002
UCG030	168570	226419	29	Dunkellin	Rafford	Cloghervaun R.	NE Brackloon	2002
UCG031	166297	230498	29	Dunkellin	Rafford	Rafford	S. of Coppanagh	2002
UCG032	168885	230788	29	Dunkellin	Rafford	Rafford	d/s Woodlawn br.	2002
UCG033	168675	231452	29	Dunkellin	Rafford	Rafford	Cloonahinch	2002
UU001	271332	362390	3	Ulster Blackwate	Oona Water	Oona Water	"Site1, top=top island bside cattledrink btm=riffle bside tre"	2001
UU002	273108	358739	3	Ulster Blackwate	Oona Water	Oona Water	"Site2 topnet= at weir d/s riprap, btm = on riffle"	2001
UU003	272812	358049	3	Ulster Blackwate	Oona Water	Oona Water	"site 3, u/s of br., bottom net on riffle d/s of cattle drink"	2001
UU004	277450	357062	3	Ulster Blackwate	Oona Water	Oona Water	"Site 4, topnet=riffle bside holly tree btmnet=end of treelin"	2001
UU005	271332	362390	3	Ulster Blackwate	Oona Water	Oona Water	Site 1	2002
UU006	273009	358838	3	Ulster Blackwate	Oona Water	Oona Water	site 2	2002
UU007	272812	358246	3	Ulster Blackwate	Oona Water	Oona Water	Site 3	2002
UU008	277450	356963	3	Ulster Blackwate	Oona Water	Oona Water	Site 4	2002
UU009	275674	357161	3	Ulster Blackwate	Oona Water	Oona Water	Site 5	2002
UU010	273700	355680	3	Ulster Blackwate	Oona Water	Oona Water	Site 6	2002
UU011	272417	357358	3	Ulster Blackwate	Oona Water	Oona Water	Site 7	2002
UU012	270049	360318	3	Ulster Blackwate	Oona Water	Oona Water	Site 8	2002
UU013	270937	362884	3	Ulster Blackwate	Oona Water	Oona Water	Site 9	2002
UU014	271233	363278	3	Ulster Blackwate	Oona Water	Oona Water	Site 10	2002
UU015	275476	358542	3	Ulster Blackwate	Oona Water	Oona Water	Site 11	2002
UU016	270247	357062	3	Ulster Blackwate	Oona Water	Oona Water	Site 12	2002
UU017	277351	357851	3	Ulster Blackwate	Oona Water	Oona Water	Site 13	2002