Effects of Water Quality on Freshwater Fish Populations

WRc plc

R&D P-72

NRA
National Rivers Authority

THE EFFECTS OF WATER QUALITY ON FRESHWATER FISH POPULATIONS - FINAL REPORT

Report No: PRS 2481-M/1

November 1990

Authors: C P Mainstone and J Gulson

Contract Manager: J Seager

Contract No: 4724

Client's Reference No: 3.1.1a

This report was produced under contract to the National Rivers Authority and equivalent bodies in Scotland and Northern Ireland, and was released to the Public Domain in January 1992.

Any enquiries relating to this report should be referred to the author at the following address:

WRc plc, Henley Road, Medmenham, PO Box 16, Marlow, Buckinghamshire SL7 2HD. Telephone: Henley (0491) 571531

THE EFFECTS OF WATER QUALITY ON FRESHWATER FISH POPULATIONS - FINAL REPORT

by C P Mainstone and J Gulson

SUMMARY

There is a need to determine quantitative relationships between fishery status and water quality in order to make informed judgements concerning fishery health and the setting of environmental quality standards for fishery protection. Such relationships would also assist in the formulation of a system for classifying fisheries.

A national database of fisheries and water quality has been collated from the archives of pollution control authorities throughout the UK. A number of probable and potential water quality effects on fish populations have been identified from a thorough analysis of the database, notwithstanding large confounding effects such as habitat variation and fish mobility, and the generally sparse nature of water quality information. A number of different approaches to data analysis was utilised, and the value of each has been appraised. Recommendations concerning the integration of water quality assessment approaches have been made and further research on fishery status, and its measurement, in relation to water quality has been suggested.

Report No: PRS 2481-M/1, November 1990
117 Pages, 17 Figures, 36 Tables, 6 Appendices
Project reference: 3.1.1a

CONTENTS

CONTENTS - CONTINUED

LIST OF TABLES

List of Tables continued

LIST OF FIGURES

SECTION 1 - INTRODUCTION

This report has been produced as part of the WRc Environment Programme under project 3.1.1a, Water Quality and Fish Populations.

Although a vast number of ecotoxicological studies have been performed on fish, few studies have attempted to quantitatively relate the distribution of fish populations in river systems to the prevailing water quality. Such field studies are needed to:

i) Assist environmental managers in making operational judgements and decisions concerning the health and enhancement of fish stocks in relation to water quality management;

ii) Determine whether 'fishery status' can be used in conjunction with existing and proposed classification schemes for the assessment of surface water quality, in order to give a more comprehensive appraisal;

iii) Assist in the formulation of appropriate standards to support Environmental Quality Objectives (EQOs) concerning fishery health and general ecosystem conservation, which the National Rivers Authority (NRA) are statutorily bound to implement by 1992.

There is a need to identify and quantify the constraints acting on fish populations so that informed decisions can be made on which ameliorative measures are appropriate to any given situation.

NRA is responsible for the 1990 quinquennial River Quality Survey, and it is proposed that a biological system of classification, based on invertebrate fauna, is incorporated into this. It would therefore be desirable to investigate the relationship between fishery and biological status as assessed by invertebrate monitoring, in addition to that between fishery and water quality status.

The formulation of use-related EQOs is underway, and water quality criteria will be adopted to safeguard each of these. Field-derived relationships between fishery status and water quality are of direct relevance to the protection of salmonid and cyprinid fisheries, and water quality criteria for general ecosystem protection are also likely to be influenced by the water quality requirements of fish.

In view of the perceived needs described above, it was decided that an investigation should be conducted utilising the large historical field database accumulated by the water authorities. The agreed objectives of the study were to:

i) Establish any relationships that may exist between fish populations, water quality and biological quality from data that are currently available.

ii) Recommend future research that is required to both investigate these relationships further for the purposes of better management and to test the conclusions drawn in the study;

iii) Make recommendations as to how the conclusions of this work may be applied to the operational management of water quality and fisheries.

Objective (i) was subsequently broken down to include the following areas:

a. investigation of the relationship between National Water Council (NWC) river quality class and fishery status;

b. investigation of the relationships between specific water quality determinands and fishery status;

c. development of a computer program which combines the toxicities exerted by various water quality determinands to give an indication of total toxic effect at any given site;

d. identification of fish species indicative of waters with differing levels of pollution.

SECTION 2 - LITERATURE REVIEW

2.1 LABORATORY STUDIES

It is not within the scope of this report to review the toxicological database on fish, since the literature is extensive and has been reviewed by a number of workers; indeed, the Ecotoxicology Group at WRc are currently working on a review of the literature on toxicity to indigenous fish species.

Comprehensive published reviews include: Alabaster and Lloyd (1982), who drew together the studies made by the European Inland Fisheries Advisory Commission (EIFAC) which led to the formulation of tentative water quality criteria for the most commonly occurring toxins; Spehar et al (1980), who reviewed over 400 references; Pickering et al (1989), who tabulated the findings of an extensive review on freshwater organisms in general; and Mayer and Ellersieck (1986), who have constructed a database on the acute toxicity of 410 chemicals to 66 species of freshwater organisms.

Reviews of the toxicity to aquatic organisms (including fish) of ammonia (Seager et al 1988), chromium (Mance et al 1984a), inorganic lead (Brown et al 1984), zinc (Mance and Yates 1984a), copper (Mance et al 1984b), nickel (Mance and Yates 1984b), arsenic (Mance et al 1984c), vanadium (Mance et al 1988a), inorganic tin (Mance et al 1988b), organotins (Zabel et al 1988a), boron (Mance et al 1988c), sulphide (Mance et al 1988d), iron (Mance and Campbell 1988), mothproofing agents (Zabel et al 1988b), aluminium (O'Donnell et al 1984) and pH (Wolff et al 1988) have been published in a series of technical reports by the WRc. These form the basis of the respective UK existing and proposed Environmental Quality Standards (EQSs) (see Section 2.3).

Atchison et al (1987) have reviewed the literature on the effects of metals on fish behaviour. Behavioural tests, particularly using avoidance reactions, are often used in the US to add environmental realism to traditional laboratory toxicity testing. Avoidance tests attempt to make allowance for the ability of fish to migrate from areas of harmful water quality, if given a distinct choice. However, Lowest Observable Effect Concentrations (LOECs) are often higher than the 96hr LC50, and the laboratory-induced boundary between good and bad water quality is usually sharp - a well-defined choice which would rarely occur in the field. This said, the sensitivity of chemoreception varies between species and between toxicants, such that avoidance responses can yield environmentally relevant data at sub-lethal concentrations.

In addition, a number of other sub-lethal stress tests are in use, including growth tests, early life-stage experiments, and physiological and biochemical indicators.

2.2 FIELD STUDIES

A number of approaches have been adopted to investigate relationships between fishery status and water quality in the field. In some, the fishery of a river is investigated in relation to a particular polluting source, often upstream and downstream of a point discharge. Others are general fishery or ecological surveys of rivers in which the distribution and abundance of fish may be somewhat retrospectively compared to available water quality information. Another approach is the toxicity-based approach in which fish survey information is compared against indices of toxicity derived from laboratory toxicity testing. All of these approaches have been drawn upon in deriving water quality standards such as the EIFAC criteria (see Section 2.3).

2.2.1 Studies of particular pollution sources

There have been many investigations of the effects of pollution inputs. Examples include Herbert et al (1961) who investigated the effects of china clay wastes in three Cornish Rivers. Suspended solids were identified as being the principal variable between unpolluted and polluted sites. Suppressed brown trout (Salmo trutta L) populations were observed at the more polluted sites and fry were absent, suggesting a lack of breeding success. Though some fish at polluted sites showed evidence of gill damage and there was a reduced abundance of benthic invertebrate food available, the trout did not show a reduced growth rate.

Tsai (1970, 1971) investigated the effects of sewage effluent discharges on fish communities in Virginia, Maryland and Pennsylvania. Chlorine deriving from the practice of chlorinating effluents was thought to be a major causative agent in the degradation of fish communities below outfalls, along with increases in turbidity resulting from sewage sludge. Ammonia and detergent concentrations were not considered to be toxic. The 1971 study found that the fish species diversity index, number of species and number of fish were well correlated with each other. These parameters were negatively correlated with measurements of total chlorine, detergent, ammonia, turbidity and total phosphate only at downstream stations in both amount and increment. Conductivity, alkalinity and acidity were negatively correlated with the fish community parameters at the upstream sites, downstream sites and with their increments. Nitrite-nitrate nitrogen concentrations were negatively correlated with fishery indicators at upstream sites. Dissolved oxygen levels were positively correlated to the fishery parameters which seemed to be of use in describing fishery quality.

Many unpublished water authority reports have been produced describing fisheries in relation to identified pollution sources. However, these reports are not widely available and the water quality at sites of fish sampling is often inferred rather than being systematically measured in tandem with the fisheries data. Water authority reports on general surveys rarely link fishery status with temporally and spatially compatible water quality sampling.

2.2.2 General surveys

Learner et al (1971) and Edwards et al (1972) conducted ecological surveys of rivers in South Wales affected by industrial and domestic wastes. Learner et al (1971) studied the River Cynon, a trout stream and tributary of the River Taff, and noted discontinuities in fish populations below industrial discharges occurring in the lower river. Above the discharges, brown trout , bullhead (Cottus gobio L), eel (Anguilla anguilla L), minnow (Phoxinus phoxinus L), stickleback (Gasterosteus aculeatus L) and stoneloach (Noemacheilus barbatulus L) were recorded, though the most upstream sites and tributaries contained only bullhead and trout. In the vicinity of the discharges fish were absent, and downstream only minnow, sticklebacks and stoneloach were recorded. Pollution influences were: solids from the coal industry; raised concentrations of ammonia and biological oxygen demand (BOD); reduced dissolved oxygen concentrations; and possibly episodic concentrations of cyanide and phenol. Indications of organic pollution increased downstream. Both upper river reaches and clean tributaries showed an improved macroinvertebrate fauna when compared with the lower river and seemed to be greatly affected by inputs of coal solids.

Edwards et al (1972) carried out a broader survey of the River Taff. Apart from domestic and coal industry wastes the river also received a highly organic effluent from a gelatin-making factory in the lower river which showed reduced dissolved oxygen concentrations. Both the main river and the Rhondda, a major tributary, were found to have high concentrations of suspended solids and BOD, though dissolved oxygen levels were not significantly reduced. Though measured by the river authority, ammonia data were not presented and presumably it was not considered to be a significant toxicant in the system. Bullheads and trout were restricted to the upper river and tributaries. The authors considered bullheads particularly sensitive to solids, this species being absent from sites at which the average concentration of suspended solids exceeded 20 mg l^{-1}. Stoneloach and minnows were widely distributed. Sticklebacks were widely but patchily distributed. Lampreys (Lampetra planeri) were only recorded at a single tributary site and eels were abundant, though restricted to the lower river. Weirs restrict the passage of migratory fish, such as eels, through the system. Solids were considered to be limiting trout through spawning success, though concentrations of copper and zinc in the Rhondda were also calculated (using methods described in Section 2.2.3) to be sufficiently toxic to restrict fish populations. The macroinvertebrate fauna was found to be impoverished at polluted sites. Gammarus was notably absent from much of the catchment; this genus was considered to be sensitive to suspended material by Hynes (1960). Asellus aquaticus was more widely distributed. Oligochaetes and chironomid larvae were abundant at polluted sites and these invertebrate taxa are typically common in organically-enriched waters with silty substrates.

Williams and Harcup (1974) continued investigations on the industrial rivers of South Wales with a study of the fish populations of the River Sirhowy, a major tributary of the River Ebbw. The river, a trout river by nature of flow regime and habitat, was affected by high levels of suspended solids derived from the coal mining industry. Trout were widely distributed throughout the river but this was partially attributable to stocking of trout by angling clubs. Reproductive success was limited to some (cleaner) tributaries. Stoneloach and minnows were present throughout the main river and also occurred in some tributaries. Sticklebacks were also found throughout the river system as occasional specimens. Eels were only found in the lower river, probably reflecting access problems from the sea via the polluted Ebbw Fawr. Bullheads were restricted to tributaries and to small numbers in the upper river. Small numbers of roach (Rutilus rutilus L) and chub (Leuciscus cephalus L) were found in the lower river.

A tributary of the River Taff, the Taff Bargoed, was the subject of invertebrate and fish studies by Scullion and Edwards (1980a, 1980b). The trout stream was polluted in its headwaters by acid drainage from coal stockpiles, by coal mine-water discharges in its middle reaches, and by ferruginous drainage in a tributary. Trout densities were very low downstream of the acidic and ferruginous discharges and were reduced in sections of the river with average suspended solids concentrations of

100 mg l^{-1}. Feeding of the trout was affected in the sites polluted by suspended solids, fish stomach contents containing a major increase in terrestrial invertebrates. Egg survival studies demonstrated reduced survival of eggs buried in the substrate at sites affected by both solids and ferruginous drainage. The acid-affected reach from which trout were absent was not subject to such studies. Stoneloach were widely distributed, though at a reduced density at sites affected by solids pollution. Only a single bullhead was found, suggesting sensitivity to both solids, acidity and possibly to ferruginous drainage. In an earlier study (Edwards et al 1972) carried out prior to extension of the coal stock-piles in the headwaters, the species had a shown a more extensive distribution. Minnows were present in the upper and middle reaches and eels were present in the middle and lower reaches. Sticklebacks were recorded in the tributary affected by ferruginous drainage. The presence of minnows in the most acid-affected reach was thought to be more a result of an ability to rapidly recolonize sites rather than, necessarily, a tolerance to pollution. Invertebrate communities were affected by the three pollution sources.

Studies on the long-term effects of road construction on the ecology of a Canadian stream (Taylor and Roff 1986) found that there was a decrease in the numbers of bottom feeders (including mottled sculpin, Cottus bairdi) with an increase of midwater fish during the period of increased sediment deposition. Populations of bottom-feeding fish and the sediment-affected invertebrate community had recovered somewhat five years after the completion of the road.

Van Loon and Beamish (1977) investigated the effects of heavy metal contamination, principally by zinc and copper, on the water chemistry and fish populations of some Canadian lakes. Populations of fish were found in lakes with a mean zinc concentration of 0.3 mg l^{-1} (calcium 16 mg l^{-1}, magnesium 3 mg l^{-1}), a zinc concentration known to have significant sublethal effects. However, yellow perch (Perca flavescens) and northern pike (Esox lucius) were apparently more successful than white sucker (Catostomus commersoni), walleye (Stizostedion vitreum) and lake whitefish (Coregonus clupeaformis), which were found in low numbers. The spawning success of the suckers appeared to be impaired. Populations of fish in lakes with zinc and copper concentrations of up to 0.09 and 0.01 mg l^{-1} respectively seemed unaffected by the metals.

Mine drainage and resulting ferric hydroxide deposition was considered to reduce the standing crop of fish in a study in Pennsylvania, USA (Letterman and Mitsch 1978). Benthic fish such as sculpins (cf bullhead) were particularly affected, as was the biomass of invertebrates in the area of iron deposition. A similar study on the River Don in Lancashire (Greenfield and Ireland 1978) demonstrated problems of iron hydroxide originating from coal mine spoils. Bullheads and minnows were not found at or below a site of moderate pollution and further downstream. Sticklebacks and stoneloach were found at this site and also upstream along with the latter species. At heavily polluted sites no fish were found. Only sticklebacks and stoneloach were found further downstream below the confluence with the River Calder. Macroinvertebrate diversity was reduced by pollution, the upstream community consisting of oligochaetes, plecopterans, ephemeropterans, trichopterans, coleopterans, dipterans and molluscs and downstream consisting of oligochaetes and chironomid larvae. Minnows were apparently more sensitive than stoneloach in cage experiments in the River Don and a fishless tributary.

Overrein et al (1981) studied the effects of acidification in Norway and presented a list of fish sensitivity based on field observations supplemented by tank experiments. The species considered most sensitive was the rainbow trout (Oncorhynchus mykiss Walbaum), followed by salmon (Salmo salar L), sea trout (S. trutta L), brown trout, perch (Perca fluviatilis L), char (Salvelinus alpinus L), brook trout (Salvelinus fontinalis Mitchill), pike (Esox lucius L) and eels. The relative importance of pH and aluminium was not stated in this listing.

Stoner et al (1984) concluded from studies of fisheries in the acidified upper Tywi catchment that salmonid fisheries would be at risk where combinations of pH <5.4, dissolved calcium <100 $\mu eq l^{-1}$ and dissolved aluminium >20 $\mu eq\ l^{-1}$ frequently occurred. Welsh Water (1986) conducted a regional survey to assess the impacts of acidification on fisheries. A total of 90 sites were sampled in 15 major catchments on streams thought vulnerable to acidification. Quantitative fish stock assessments were made along with water and habitat quality analysis. Only 49 of the sites were thought accessible to migratory salmonids. Trout were absent from 10 of the 90 sites and were in low densities at a further 20. Salmon were found at 22 sites. Eels were found at 29 sites, minnows at 10, bullheads at 9, lampreys at 5 and stoneloach at 4. In bivariate regression analysis, pH and aluminium (log-transformed) explained significant proportions of the variance in trout abundance (39 and 41% respectively). Thus at the *deemed 'threshold'* water quality of Stoner et al (1984), a pH of 5.4 and aluminium concentration of 20 µeq l^{-1}, expected trout populations would be 7 and 9 100 m^{-2} respectively. At a pH of 6 and aluminium concentration of 8 µeq l^{-1}, levels thought to have little or no toxic effect on fish, predicted populations were 29 and 28 trout 100 m^{-2} respectively. Product-moment correlation showed that trout densities were strongly correlated with acidity factors (ie negatively with aluminium concentrations and positively with pH, and calcium and magnesium concentrations) and also (negatively) with both zinc concentration and annual daily flow (ADF). Multiple regression equations were derived using variables that were not multicolinearly related but explained a high proportion of the variance of trout abundance:

i) $\text{Log}_{10}\text{Trout Density} = 1.95 - 1.06\ \text{Log}_{10}\text{Al} - 0.285\ \text{Log}_{10}\text{ADF} - 0.718\ \text{Log}_{10}\text{Zn}$

ii) $\text{Log}_{10}\text{Trout Density} = -2.88 + 0.639\text{pH} - 0.806\ \text{Log}_{10}\text{Zn} - 0.300\ \text{Log}_{10}\text{ADF}$

These explained 52% and 54% respectively of the variance. Habitat features (except ADF) were not regarded as very important for these soft water sites, though one site was omitted from the analysis as it was considered totally unsuitable for salmonids.

Schofield and Driscoll (1987) studied the distribution of fish species in a North American catchment affected by acidification. Comparing the distribution of fish from historic data of 1931 with that of 1982, the authors found that there had been a decline in native species (eg brook trout, Salvelinus fontinalis; and slimy sculpin, Cottus cognatus), while some introduced fish species had become widely distributed. Non-native species (yellow perch, Perca flavascens; central mudminnow, Umbra limi; banded killfish, Fundulus diaphanus) were experimentally shown to be more tolerant at low pH sites than the native species in caged fish studies. High aluminium concentrations were recorded at sites showing low pH levels. A tentative classification of acid-tolerance within the system was presented, though is not reproduced here as none of the species are native to the UK.

Young (1985) conducted a statistical analysis of water quality in the Anglian Region for the evaluation of the EQS for ammonia. Chemical and fish biomass data from 285 sites were subject to statistical analysis. The study demonstrated no significant correlation between fish biomass and ammonia concentrations, though there was considerable scatter in the fish data, probably due to habitat variability and fish movements (see Section 4.2).

Cowx and Broughton (1986) studying changes in the species composition of anglers catches in the River Trent over the period 1969 to 1984 noted a change in dominance of fish caught from roach and dace (Leuciscus leuciscus L) to that of bream (Abramis brama L), chub, perch and eels. The authors suggested that water quality improvements in the river were largely responsible for these changes and presented chemical data showing declines in BOD, suspended solids and ammonia concentrations in the river. The changing techniques of anglers were also suggested as a contributory factor.

2.2.3 Toxicity-based Studies

Herbert et al (1965) extended laboratory investigations of common toxicants to study the toxicity of three river waters in the Midlands where fish were believed to be absent. The rivers were affected by sewage effluents resulting in low dissolved oxygen levels and high levels of toxicants such as ammonia, nickel, zinc, copper, chromium and cyanide. Mortality rates of trout in aerated river water were studied in comparison to predicted toxicities based on laboratory studies. Copper and zinc were particularly significant toxicants in the studies, which demonstrated a reasonable agreement between the predicted and observed toxicities of the river water.

Brown (1968) presented a method of calculating the acute toxicity to trout of mixtures of the (then) common industrial pollutants ammonia, phenol, zinc, copper, cadmium, lead, nickel and hydrogen cyanide. The technique included adjustments to the toxicity of these substances based on their known interactions with pH, hardness, alkalinity, dissolved solids and the dissolved oxygen concentration of the dilution water. The method relied on the available data on toxicity of these substances to rainbow trout and produced a predicted toxicity value as a fraction of the 48h LC50. The method assumed an additive effect of the toxicants. The use of the method in assessing the toxicity of surface waters also assumes the extrapolation of the degree of sensitivity from the laboratory to the field and also that the sensitivity of rainbow trout bears some relation to the sensitivity of naturally occurring species.

Brown et al (1970) carried out a similar study to that of Herbert et al (1965) using a range of fish species (rainbow and brown trout, roach and dace) at a range of river sites (Rivers Erewash, Don, and Tees and Billingham Beck), including an estuary site (Tees). Fish were exposed to river waters in cages in situ and in aerated aquaria. Expected toxicities to trout were estimated from the method of Brown (1968). Predicted toxicities based on concentrations of molecular cyanide, copper, zinc, ammonia and phenol were close to those found. However, generally in both the studies of Brown et al (1970) and their analysis of the data of Herbert et al (1965) the observed toxicities were greater than those predicted, the observed 48h LC50 being between 0.6 and 0.7 rather than 1.0 times the predicted LC50. There was a greater underestimation of values from saline waters, the LC50 values being about 0.3-0.4 of the predicted values. The value of applying toxicity predictions in these instances is therefore reduced. The difference between predicted and observed results would probably have been reduced if more toxicants could have been included in the predictive calculations. Other contributory factors were interactive effects of toxicants on the fish (estimations of toxicity presume an additive effect) and also the effects of fluctuating toxicant concentrations, which were averaged for predictive calculations. Brown et al (1970) found coarse fish more resistant than trout, with dace being more resistant than roach.

Alabaster et al (1972) extended the toxicity approach and presented plots of the proportion of 48h LC50s against the cumulative percentage of samples taken from a particular river from both fishless sites and those where fish were present. The toxicity predictions excluded the effects of low dissolved oxygen levels and also of non-assayed substances such as detergents, chlorinated hydrocarbons and suspended solids, but there was a reasonable agreement between the state of fisheries and the predicted toxicities. Predicted toxicity was calculated as the sum of estimated fractions of the 48h LC50 for trout of concentrations of soluble copper, zinc, phenol, ammonia and cyanide. In the Trent catchment, for sites reported to have both trout and coarse fish, the predicted toxicity was always less than 0.3 with a median value of 0.1. At fishless sites, though the sum was less than 0.3 for 55% of the sites the median was about 0.25. Further samples taken in 1968 and 1969 on the Trent included additional analysis for nickel and cadmium. The distributions of predicted toxicities for 1968 showed median toxicities of 0.1 for game fisheries and 0.2 for coarse fisheries. The corresponding values for 1969 were 0.03 and 0.12 respectively. In 1968 the division between fishless and fish-supporting waters was demonstrated at a median toxicity of 0.28, with 5 and 95 percentiles (95%ile) of 0.1 and 0.73 respectively. Consideration of dissolved oxygen concentrations indicated that only slightly raised concentrations may have enabled fish survival at some marginally fishless sites. The contribution to the toxicity of nickel, chromium and cadmium was underestimated in the predictions based on 48h LC50 values as these toxicants are all demonstrably toxic to fish in the long term. Recalculations based on median threshold concentrations increased both the contributions of these and total metals to the overall toxicity. Studies on the Willow Brook demonstrated a predicted median toxicity value for the boundary between fishless and fish-supporting sites of between 0.32 and 0.25, comparing well with the value of 0.28 for the River Trent.

Studies on the Willow Brook were continued by Solbe (1973). The stream was affected by mining (iron ore), and industrial and sewage effluents with high ammonia, phenol and low dissolved oxygen concentrations being recorded at times. Zinc and anionic detergents were also consistently recorded at significant concentrations, though the detergent residues were presumed to be relatively harmless. Iron, copper, lead, cadmium, chromium, nickel and mercury were also present. In general, however, only zinc, ammonia and to a lesser extent, copper and lead contributed to the predictions of toxicity. Considerable day-to-day fluctuations of predicted toxicity were shown at sites, though overall values and fluctuations decreased further downstream of discharges due to dilution, attenuation and the buffering effect of lakes in the system. Distribution plots of predicted toxicity demonstrated significant departures from a straightforward log-normal plot at a high (cumulative) percentage occurrence, such that predicted toxicities exceeded those expected from extrapolation of the data up to a given level of occurrence. It was thought, however, that exposures to potentially lethal concentrations for short periods of time could be tolerated by fish, such that a fishery could be maintained. The site showing the highest median and 95%ile toxicity values (0.45 and 2.35 respectively) was fishless. Fisheries improved downstream with decreasing median toxicity, a good mixed fishery occurring at the most downstream site where the median predicted toxicity was 0.17 and the 95%ile value was 0.62. Roach predominated at some of the more upstream of polluted sites, though sticklebacks predominated at the most impacted site at which fish were present. Stoneloach were suggested as being a sensitive species, being absent after a period of poor water quality. Dace, chub and minnow were thought more sensitive than roach, tench (Tinca tinca L) and gudgeon (Gobio gobio L) as the latter species existed in waters having a median toxicity of over 0.34 of the predicted 48h LC50 to rainbow trout whereas the former were rarely found penetrating the brook upstream of a reach having a median toxicity value of 0.17. Laboratory studies were later carried out on the toxicity of zinc and cadmium to rainbow trout and stoneloach to investigate the distribution of stoneloach in the Willow Brook (Solbe and Flook 1975). Stoneloach were found to be more sensitive than trout to zinc but much less sensitive to cadmium toxicity, suggesting that the absence of the fish in the latter part of the study may be explained by zinc toxicity. A further study of Willow Brook (Solbe 1977) demonstrated that invertebrate diversity and biotic indices were found to increase downstream with improving water quality, and although this improvement was similar to the change of status of the fishery it was not possible to predict the fish community on the basis of biotic indices alone.

Solbe and Cooper (1975) studied the fisheries of the River Churnet in relation to the predicted toxicity of the river water. The river was polluted by sewage and trade wastes, leading to low dissolved oxygen concentrations at some sites and, more generally, to high concentrations of copper, zinc and lead. Again at sites with a high predicted toxicity largely due to copper, fisheries were severely affected. Trout and bullhead were found in both upstream and tributary sites. Sticklebacks, minnow, pike and stoneloach were present at a small number of sites, including the most downstream site, accessible from the River Dove. The same authors conducted work on the River Tean (Cooper and Solbe 1978). The effluent from a sewage treatment works (STW) receiving domestic and industrial wastes affected the water quality of the river system. Compared to the single water quality sampling site above the discharge, the two sites downstream showed raised concentrations of hardness, alkalinity, conductivity, dissolved solids, oxidised nitrogen, soluble phosphorus, BOD, and the metals cadmium, chromium, nickel, lead, copper and zinc. Median predictions of toxicity (as a proportion of the 48 hour LC50) to rainbow trout at the three sites did not exceed 0.12, and 95%ile toxicities did not exceed 0.225. However, the toxicity of cadmium was considered to be underestimated using the short-term 48h LC50 approach, and estimations based on known lethal threshold concentrations indicated the fraction of threshold toxicity exceeded unity due to cadmium alone at the middle site which was most influenced by the discharge. Undiluted effluent was lethal to rainbow trout in fourteen days. Fish were present throughout the River Tean (eleven sampling sites) though trout were absent below the sewage works, except for a single fish which may have been introduced. The limited trout distribution may have been attributable to cadmium toxicity. Other, probably less sensitive, species (bullhead, stickleback, minnow and stoneloach) were more widely distributed, though stoneloach were not found in the vicinity of the STW.

Howells et al (1983) investigated water quality, fish and invertebrates in the Mawddach river system, Wales. Water quality problems included copper, zinc and iron from natural outcrops and abandoned mines and reduced pH values. Predicted toxicities of copper and zinc were calculated and the median fractions of the threshold LC50 to rainbow trout exceeded 0.4 in two tributaries. The Mawddach headwaters at times showed low pH values with 95%iles of less than 4.75. Trout were widely distributed throughout the system as were eels. Minnows were found in one of the polluted tributaries. The trout caught included sea trout smolts at some downstream sites in the system. The biomass of trout populations at each site was closely related to predicted toxicity values. Number and biomass of trout were reduced to zero at predicted threshold LC50 values of about 0.6 (median) and 1.3 (95 percentile). Low pH was considered to be a contributory factor to reduced stocks in the Mawddach headwaters. Aluminium concentrations were not measured but were presumed to be low. Fish biomass was found to be unrelated to invertebrate biomass. The distribution of Ephemeroptera was related to the recorded pH and hardness (though not metal concentrations); few of these invertebrates were found in the Mawddach.

Pihan and Landragin (1985) extended the approach of Alabaster et al (1972) in a study of the fisheries of French rivers showing varying degrees of pollution. Cumulative percentage plots of predicted toxicity indicated fishless sites would be found to have median and 95%ile toxicities exceeding 0.8 and 1.75 times the 48 hour LC50 to rainbow trout. 'Good' fisheries would be expected where predicted median and 95%ile toxicities were less than 0.3 and 0.7 respectively. Between these median and 95%ile values, intermediate and perturbed fisheries were expected. The distribution of data from two rivers was very skewed above a percentile value of around 98 indicating that the use of distribution up to and including the 95%ile value would severely underestimate the toxicity at more extreme values. This demonstrated that even the use of 95%iles as a measure of extremes may be inadequate.

In another investigation of the effects of acidity and other factors, Turnpenny et al (1987) studied the fish populations of some streams in Wales and northern England. Of 60 streams sampled, 45 contained fish. All contained trout, 13 contained eels, 6 salmon, 6 minnow, 2 bullheads and one site contained lampreys. The more acidic sites showed fewer fish species and reduced densities. Acidic sites showed raised concentrations of trace metals (Al,Cu,Zn and Pb), and the absence or reduced density of trout (and salmon) was related to high monomeric aluminium concentrations (>40 ugl^{-1}) and predicted toxicity values (>0.4 - see Section 2.2.3) of copper, zinc and lead combined (based on the threshold LC50 to rainbow trout). The effects of water quality were thought to be direct rather than secondary effects through the food chain. The authors categorised the sites according to criteria thought only applicable to waters of similar water chemistry (ie mean pH>5.0, calcium<1mg l^{-1}):

Good fisheries:	Al<40 ugl^{-1}, Cu-Pb-Zn toxicity <0.4 tLC50
Moderate fisheries:	Al<40 ugl^{-1}, Cu-Pb-Zn toxicity 0.4-0.7 tLC50
Poor fisheries or fish absent:	Al>40 ugl^{-1}, Cu-Pb-Zn toxicity >0.7 tLC50

2.3 FISHERIES CLASSIFICATION AND ENVIRONMENTAL STANDARDS

The Report of the 1970 UK River Pollution Survey (DOE 1970) included fisheries considerations in river classifications. Class A rivers were regarded as those with good game or mixed fisheries. Class B rivers were regarded as those with good mixed fisheries. Class C rivers were regarded as those with moderate to poor fisheries, with fish populations being mainly restricted to roach and gudgeon. Class D rivers were those known to be incapable of supporting fish life. Such consideration of fisheries was not included in later surveys. The Association of River Authorities produced a working party report (Tombleson 1974) on coarse fisheries which included estimated limits of water quality to support types of coarse fishery based on Trent River Authority data (Table 1). The derivation of these limits was not explained but as they are relatively high one assumes that they were set on the broad bands of water quality in which the types of fish were found.

Table 1 - Limits of quality to support types of coarse fisheries

(after Tombleson 1974)

Fishery	BOD mg l^{-1}	Ammonia mg l^{-1}	Dissolved oxygen mg l^{-1}	Temperature °C
Grayling/Trout	<3.3	<0.5	>6.9	<20
Chub/Dace	<5.0	<0.9	>5.0	<28
Roach/Gudgeon	<9.5	<3.3	>2.0	<30
No Fish	>9.5	>3.3	<2.0	>30

UK Environmental Quality Standards pertinent to the protection of freshwater fish are outlined in Appendix A.1. These are derived from the European Council Directives 78/659/EEC (the 'freshwater fish' directive - CEC 1978) and 76/464/EEC (the 'dangerous substances' directive - CEC 1976). Those standards derived from the former directive only apply to designated (either salmonid or cyprinid) river stretches; no national standards have yet been adopted for some of these parameters (eg DO, suspended solids). However, NRA regions may have adopted their own environmental standards for the protection of fisheries and other freshwater life.

EIFAC has developed tentative water quality criteria for the protection of freshwater fish, concerning commonly occurring toxins (Alabaster and Lloyd 1982).

The US Environmental Protection Agency (EPA) periodically review ecotoxicological information (EPA 1986) pertinent to the extensive list of EPA water quality criteria, given in Appendix A.2, and update these criteria as necessary.

SECTION 3 - ANALYSIS OF NATIONAL DATABASE

3.1 DATA COLLATION

3.1.1 Data sources

Information concerning fish and biological (invertebrate) surveys, and routine water quality monitoring sites, was gathered from the NRA regions. In Scotland, the River Purification Boards (RPBs) provided invertebrate and water quality data, whilst the Department of Agriculture and Fisheries for Scotland (DAFS) provided fisheries information. Water quality and invertebrate data were also provided by the Department of the Environment (Northern Ireland) (DoE(NI)), whilst the Department of Agriculture for Northern Ireland (DANI) provided fisheries information.

3.1.2 Site selection

An effort was made to cover a wide range of water quality and habitat types. In addition, the following site selection criteria were generally adhered to:

i) the fish, water quality and invertebrate monitoring sites (termed the 'linked sites') all lie within approximately 1km of each other (the site is not rejected if a linked invertebrate monitoring site is absent);

ii) there is no interference between linked sites from significant tributaries, weirs or (as far as is known) effluent discharges;

iii) the linked sites do not straddle NWC class change, as identified by the appropriate quinquennial River Quality Survey map;

iv) a minimum of 6 routine water quality samples were taken in the 12 months preceding the date of the fish survey, each comprising at least a basic sanitary analysis;

v) fish data are available as true estimates of both density and biomass on a species-by-species basis, preferably including the minor species;

vi) the date of the invertebrate survey lies (approximately) within the 12 months preceding the date of the fish survey (if not, the linked invertebrate monitoring site is rejected).

Ninety-one sites were selected from the information base supplied, of which 13 have 2 datasets, 1 has 3 datasets and 2 have 4 datasets, due to repeat fish surveys. This results in a grand total of 112 datasets. However, of the 91 sites, only 45 (56 out of 112 datasets) had linked biological (invertebrate) monitoring sites. A regional breakdown of the sites is given in Table 2, and full details are given in Appendix B.

In using site selection criterion (v), which was included in order to produce a database that was fully quantitative and as compatible as possible, a significant proportion of potential sites has had to be omitted. Apart from criterion (v), a large number of sites was also rejected on the basis of criterion (i) (including all sites in Northern Ireland on the River Bush), the other criteria being of less importance.

Table 2 - Regional breakdown of the database

Region	Sites	(Linked) (invert) (sites)	Repeat fish surveys	(Repeat) (invert) (surveys)
NRA Anglian	24	(14)	1	(0)
NRA North West	4	(0)	0	(0)
NRA Severn Trent	24	(17)	17	(9)
NRA South West	9	(1)	0	(0)
NRA Southern	11	(2)	1	(0)
NRA Thames	8	(5)	1	(1)
Tweed RPB	9	(5)	1	(1)
Forth RPB	2	(1)	0	(0)
	91	(45)	21	(11)

3.1.3 Parameter selection

Fisheries

A number of parameters can be used to assess fishery status. Population density and biomass, relative abundance and presence/absence are the most frequently used within the NRA, but other useful characteristics include growth rate, production, species diversity, age structure, body condition and even trophic structure.

Growth rate and production have the advantage that they measure dynamic processes which relate to water quality over any desired timescale; the main disadvantage is that they are density-dependent, such that an idea of carrying capacity is required for proper interpretation (Zalewski et al 1985). Species diversity, either in its simplest form of 'number of species' or as a diversity index incorporating abundance, has often been used to indicate fish community degradation (eg Portt et al 1986; Tsai 1970; Lelek 1981). Age structure can identify the timing of significant events affecting fish populations, but cannot readily distinguish them from natural fluctuations in year class strength. Body condition gives an indication of the degree of stress to which individual fish are subjected. Trophic structure can be used to show

how fish populations are thrown out of equilibrium by environmental perturbation, and is a fundamental constituent of the Index of Biotic Integrity (IBI) (Karr 1981), in association with species composition, abundance and fish health (condition). The IBI is increasingly used in the US as an indicator of habitat degradation.

Of the fisheries characteristics mentioned above, density, biomass and species composition (presence/absence and species number) were available at nearly all sites in the database. Density and biomass are broken down to the species level, but at a proportion of sites only presence/absence or no data are available for the minor species. All of these parameters have been utilised in the data analysis.

Water Quality

A list of desired parameters for inclusion in the data analysis was formulated from the toxicological literature (see Appendix C), but inevitably parameter selection was largely controlled by data availability. It was decided to use the water quality data for the 12 months prior to the date of the fish survey at each site, and a determinand was rejected at any given site if the frequency of analysis was below 6 per annum. Appendix C shows the frequency of occurrence of each desired water quality parameter within the database. The appendix includes some parameters which are not toxic to fish (eg hardness, conductivity) but which have a bearing on the toxicity of other parameters.

For the majority of sites only a basic sanitary suite was available, comprising:

Temperature
pH
Dissolved oxygen (DO)
Total Ammonia (NH_3N)

Biochemical Oxygen Demand (BOD)
Suspended Solids (SSLDS)
Nitrite (NO_2N)
Nitrate (NO_3N)
Phosphate (PO_4)

In addition, metal analyses (usually 'total' but occasionally 'dissolved') may have been performed, which would normally comprise the following:

Zinc (Zn)
Chromium (Cr)
Nickel (Ni)
Copper (Cu)
Cadmium (Cd)
Lead (Pb)

Aluminium (Al), Arsenic (As) and Iron (Fe) occur rarely in the database. Cyanide (CN), synthetic detergents, and all types of organic determinand either do not occur or occur at a frequency lower than 6 samples per annum; for this reason, these determinands were not considered in the data analysis.

The NWC class of each site was taken from the regional maps of the quinqennial NWC River Quality Survey closest to the date of the fishery survey. It was decided to assess NWC class in this way because:

i) unless an NRA region produces interim NWC class assessments, the quinqennial maps are the most current assessment available to environmental managers;

ii) as the criteria used for NWC class assessment varies between NRA regions, a separate assessment based on the water quality data used in this study would produce a measure of standardisation which in reality does not exist.

No NWC class is ascribed to sites in Scotland, since a different classification scheme is used there.

Habitat

Habitat is one of the major determinants of the size and nature of fish populations, upon which the effects of water quality are superimposed. It is important, therefore, to consider habitat as fully as possible in a study such as this, where fish populations from very different habitats are being examined within the same database. Milner et al (1985) give an indication of the habitat characteristics that influence fish distribution, and the subject is discussed further in Section 3.2.2.

Unfortunately, habitat characteristics are inconsistently observed (between regions) during fish surveys, and when they are observed there is no UK-wide (or even NRA-wide) standardisation on the specific parameters to be reported. Channel width is the only regularly recorded habitat parameter, with water depth (either as a mean or a maximum) being recorded in some regions but not in others. Channel width has been found to be a useful habitat parameter (Huet 1959), mainly due to its relationship with flow (Pitwell 1976). Water depth may indicate a potentially restrictive habitat (shallow water) or areas of reduced DO (deep water). Where water depth was recorded as a maximum value it was converted to a mean depth by applying a factor of 0.67, on the basis of the relationship between maximum and mean depth in the cases where both were available.

Observations on substrate composition and vegetation cover are recorded in some regions, but not in a manner consistent between regions. For this reason these parameters have not been included in the database.

The hydrological regime (flow, current velocity) has a great bearing on fish distribution, and must be taken into account in some way. Current velocity determines substrate composition and influences natural longitudinal gradients in dissolved oxygen and temperature, apart from having a direct effect on river flora and fauna. Since current velocity is only available from perhaps one or two gauging sites in any one catchment, slope has been used as a surrogate parameter. Slope is directly related to current velocity and flow rate (Pitwell 1976) and may conveniently be taken from Ordnance Survey (OS) maps. The combination of channel width and slope give an even better representation of flow (Persoone 1979). Slope, along with altitude, was taken from 1:50 000 OS maps.

Appendix C shows the frequency of occurrence of each habitat parameter within the database.

Invertebrates

Although invertebrate biomass or density would probably be a more relevant parameter in relation to fishery status (although this would not account for prey preference), invertebrate data is generally recorded on a presence/absence basis and formulated into a biotic score. Different biotic scores have found favour in different regions (eg the Trent Biotic Index in NRA Severn Trent Region and the Lincoln Quality Index in NRA Anglian Region), but all regions calculate the Biological Monitoring Working Party (BMWP) Score and some calculate the associated Average Score Per Taxon (ASPT). Since these are the only invertebrate scoring systems that have been used on a widespread basis and are thus available at most sites, the BMWP score and ASPT have been adopted in this study.

Appendix C shows the frequency of occurrence of the BMWP score and ASPT within the dataset.

3.2 LIMITATIONS OF THE APPROACH

3.2.1 Database constraints

Fisheries data compatibility

Variations in the efficiency of capture of smaller individuals due to differences in capture techniques means that comparison of fish density data between regions (and within regions to some extent) may not be justified. For the purpose of the data analysis it is assumed that these smaller individuals do not constitute a significant proportion of the population biomass, such that comparison of biomass data between regions is justified; the validity of this assumption will depend upon the age structure of the population, such that there will be a tendency to under-estimate the biomass of 'young' populations.

Management priorities in some regions result in the 'minor' species either not being recorded or being recorded only on a qualitative basis. In regions where migratory salmonids dominate, emphasis is usually placed on fish density, since it is the number of immature individuals available to participate in migration that is important rather than their weight.

A number of methods of population estimation have been used within the database, involving both catch-depletion (eg Zippin 1958, Seber and Le Cren 1967) and mark-recapture techniques. It has been assumed that the discrepancies introduced into the database by the use of these various methods are not significant.

Seasonal movements of fish (see Section 3.2.2) associated with spawning and winter aggregation have implications for the comparability of population estimates made at different times of the year. Ideally, the fisheries database would have been standardised to a narrow time window but, owing to the high number of site rejections based on other site selection criteria, this was not possible.

Water quality characterisation

An advantage of pollution monitoring using biological systems rather than a direct chemical assessment of water quality is that organisms will respond to toxins in the water irrespective of whether they are looked for analytically (Hellawell 1986). This means that when attempting to relate chemically monitored water quality to biologically monitored water quality, relationships will be obscured where toxins occurring in significant concentrations are not analysed for. This is very likely to be the case with the restricted water quality database used in this study (see Section 3.1.3).

Furthermore, organisms respond to the combination of all toxins present, and their synergistic, antagonistic or additive efffects. Little work has been performed on the combined effects of toxins, making environmentally realistic characterisation of water quality difficult.

Discrete chemical sampling has a further disadvantage over biological monitoring in that it is unlikely to detect infrequent but ecologically significant pollution episodes. This means that the water quality database used in this study may miss real relationships between water quality and fisheries due to an inadequate sampling regime.

For the purpose of analysis the water quality data collected in this study needed to be summarised in an environmentally meaningful way. Measures of central tendency, such as the mean or median, may be used, or emphasis can be placed on extreme events by using an upper percentile (traditionally the 95%ile). Alternatively, synoptic (in relation to the fish survey) assessments of water quality may be used. In reality, the statistic that best characterises the water quality at a particular site will depend upon local circumstances which will vary between sites.

In areas of relatively uniform water quality, a measure of central tendency will best characterise ambient water quality conditions. However, at a site where water quality fluctuates and the fish population is largely isolated from other populations by barriers such as weirs, recovery of that population following a fish kill will be slow. This means that an upper percentile is likely to best represent the water quality at that site (unless restocking is undertaken - see Section 3.2.2). Conversely, at a site where potentially recolonising populations are in close proximity and have free access, the effect of a fish kill on fish catches is likely to be short-lived. In this case, a measure of central tendency is likely to best represent water quality. In regions of heterogeneous and fluctuating water quality, fish populations may intermittently migrate into and out of an area if avoidance reactions are triggered; synoptic measurements of water quality would be most appropriate in such cases.

Any summary statistic chosen will inevitably be a compromise which best represents the database as a whole rather than any individual site.

3.2.2 Confounding field effects

Henderson (1985) divides the factors determining fish assemblages into 4 groups:

i) dispersive factors (concerning the arrival of fish at a given site);

ii) autecological factors (concerning the physiological constraints to species distribution);

iii) synecological factors (concerning relationships between organisms, including competition and predation);

iv) stochastic factors (concerning natural variability).

For the purposes of a study which attempts to quantitatively relate fishery status to water quality in heavily manipulated aquatic systems, it is necessary to modify Henderson's groupings. Group (i) must include not only dispersive factors but also the tendency of different species to range, migrate or shoal, which will cause the population size at any given site to vary with time; this group can be renamed 'fish movements'. From group (ii), those autecological constraints which are caused by anthropogenic alterations of water quality (ie pollution) must be extracted, such that only habitat constraints on species distribution are included; this allows all effects attributable to pollution to be determined. Lastly, to the 4 groups should be added two more: v) fish stocking activities; and vi) acclimation to elevated toxicant levels, acting at either the population level (genetically) or the individual level (physiologically).

These 6 groups can be seen as those factors which act to obscure the relationship between water quality and fishery status, and will now be discussed in turn.

Fish movements

The colonising potential of a species, given suitable habitat, and its population size (in terms of density and biomass) at any given time, are both largely dependent upon:

i) mobility and the tendency to range/migrate;

ii) the possible existence of barriers to fish movement.

Due to these factors some species have a patchy geographical distribution, especially on a catchment scale (since movement across watersheds is extremely difficult); such distributions will be largely dictated by local stocking activities (see Section 3.2.2) if the species are popular with anglers.

Early workers suggested that within catchments many coarse species moved about within a very limited area, termed the 'home range' (Gerking 1953). Later mark/recapture studies have indicated that species such as gudgeon, roach (Stott 1967) and perch (Bruylants et al 1986) have mobile and static components to the population. Stott (1967) suggested that such mobile components are likely to be important in the rapid recolonisation of sites with depleted populations following catastrophic events. Bruylants et al (1986) found that the mobility of perch populations increases with increased uniformity of habitat. Linfield (1985) suggests that there is a tendency for older dace and chub to move upstream that compensates for the downstream drift of fry.

Winter aggregation is commonly observed in some species (Jordan and Wortley 1985, Hynes 1979), particularly cyprinids, probably associated with movement into deeper water. Jordan and Wortley (1985) found winter aggregations of up to $1787gm^{-2}$ in the Norfolk Broads, and concluded that in this area at least, accurate population estimates can only be gained in the summer when fish are maximally dispersed.

Superimposed upon these general fish movements are spawning migrations, in which many species will move upstream, spawn and subsequently disperse downstream (Hynes 1979). A number of cyprinids (including chub, dace and barbel, Barbus barbus L), and also non-migratory brown trout, are known to undertake such spawning movements, often taking the fish into small tributaries. Lelek (1981) estimated that a river stretch of 10 - 15 km, preferably with an inflowing tributary, is required to satisfy the complete life cycle of the barbel.

The scale of such fish movements will depend upon the geographical extent of the autecological range, the presence of barriers and, in the case of general fish movements, is also likely to depend upon the uniformity of habitat. Barriers to fish movement can take a physical form, such as weirs, waterfalls and depleted flows, or may be due to stretches of poor water quality; any of these types of barrier may be seasonal. Linfield (1985) observed that a species will not occur naturally above an impassable weir if its fry cannot hold station above that weir. At a catchment level, watersheds are an efficient barrier to fish movement.

Species mobility is taken to the extreme in the case of migratory salmonids, where the presence of an adult at a particular site is not only dependent upon the water quality at that site, but upon the presence/absence of barriers, both physical and chemical, at all points along its migratory path. Its accrued biomass is a function of the quality of its marine feeding grounds. Clearly, adult migratory salmonids cannot be included in a study such as this, where fishery status is being related to site-specific water quality. In this case the population is taken as the pre-migratory immature individuals only; however, even this is dependent upon the adults' spawning success and therefore upon the physical and chemical constraints acting along the river's entire length. This dependence is partly compensated for by increases in the survival rate of fry at low population densities (low breeding success).

It is evident, then, that within populations of certain species, a proportion of fish are likely to range over fairly large areas unless their movements are restricted (by autecological constraints and/or physico-chemical barriers) and, unless water quality is uniform throughout their range, that such species will be subjected to a variety of water quality regimes which will confound relationships between fishery status and site-specific water quality. One compensatory factor is that in regions of particularly poor water quality, avoidance reactions (see Section 2.1) will dictate that the likelihood of capture is low compared to that of regions of water quality above the avoidance threshold. However, this is an effect which may produce transient populations in areas of fluctuating water quality, making the timing of the fish survey all-important and requiring that synoptic measurements of water quality be made if a relationship between fishery status and water quality is to be determined (see Section 3.2.1).

Those fish species that are more sedentary in nature are likely to bear a closer relationship to site-specific water quality than the more mobile species, although distribution may be patchy due to poorer powers of colonisation (as mentioned above). Such species include the bullhead and the stoneloach. Swales (1988) also found that chub exhibited a more sedentary pattern of activity than dace in the River Perry, although the spawning migratory behaviour of the chub must be borne in mind.

Autecological effects

Longitudinal gradients of a number of physico-chemical variables exist in rivers, of which current velocity, substratum, flow, temperature, dissolved oxygen, dissolved nutrients and hardness are the most ecologically significant (Hawkes 1975). Current velocity, which is itself dependent upon slope, is arguably the most important habitat variable concerning species distribution, due both to its direct effect on organisms and its influence on other habitat parameters (most notably substrate type, temperature and dissolved oxygen). In addition to the normal longitudinal gradient in current velocity, large fluctuations associated with spate flows can have a drastic effect on fish (and invertebrate) fauna, particularly fry (Milner et al 1981, Linfield 1985).

Since each fish species has specific tolerance limits for the variables mentioned above, natural longitudinal changes in species composition occur. This observation has been used to formulate river classification schemes based on 'zones', in which a number of discernable and characteristic fish (and invertebrate) species assemblages inhabit different types of river reach (eg Carpenter 1928, Huet 1959). A good account of such schemes is given in Hawkes (1975), from which a tabulated summary is taken (see Table 3).

Huet's classification differs from earlier schemes in that it quantitatively relates fish assemblages to physical variables (slope and channel width). It divides European rivers into four zones, indicated by (in order of decreasing slope) trout, grayling (Thymallus thymallus L), barbel and bream. Huet roughly characterised the species assemblage found in each zone and assessed relative abundances (see Table 4); as can be seen, much species overlap occurs between zones. Since grayling and barbel are patchily distributed in the UK these two zones are often renamed after minnow and chub respectively. The slope rule for the prediction of Huet's fish zones is shown in Figure 1. The zones are sometimes paired together to form an upper Salmonid region, consisting of the Trout and Grayling (minnow) zones, and a lower Cyprinid region, consisting of the Barbel (chub) and Bream zones.

Illies	Illies & Botosaneanu	Muller Illies Schmitz R. Fulda, Germany	Ricker Ontario streams	Harrison & Elsworth Great Berg River, S. Africa	Huet W. European rivers	Thieneman W. Europe		Carpenter (G. Britain)
(Eucrenon)	Zone I	Quellezone	—	Zone I—source	—	Quellen	Highland brooks	Head stream
(Hypocrenon)	Zone II		Spring creeks	—	—	Quellrinnsale		
Epirhithron	Zone III	Obere Salmonidenregion	Swift trout stream	Zone II—mountain torrent	Zone à Truite	Region der Bachforelle		Trout beck
Metarhithron	Zone IV	Mittlere Salmonidenregion	Slow trout stream	Zone IIIA—upper foothill				
Hyporhithron	Zone V	Untere Salmonidenregion	—	Zone IIIB—lower foothill—hard bottom zone	Zone à Ombre	Region der Asche		Minnow reach
Epipotamon	Zone VI	Bargenregion	Warm rivers	Zone IV—lower foothill soft bottom	Zone à Barbeau	Barbenregion	Lowland courses	Upper reach
Metapotamon	Zone VII	—	—	Zone V—flood plain	Zone à Breme	Brassenregion		Lower reach
Hypopotamon	—	—	—		—	Brackwasser-region		Brackish estuary

Table 3. Comparison of river zone classification schemes (after Hawkes 1975).

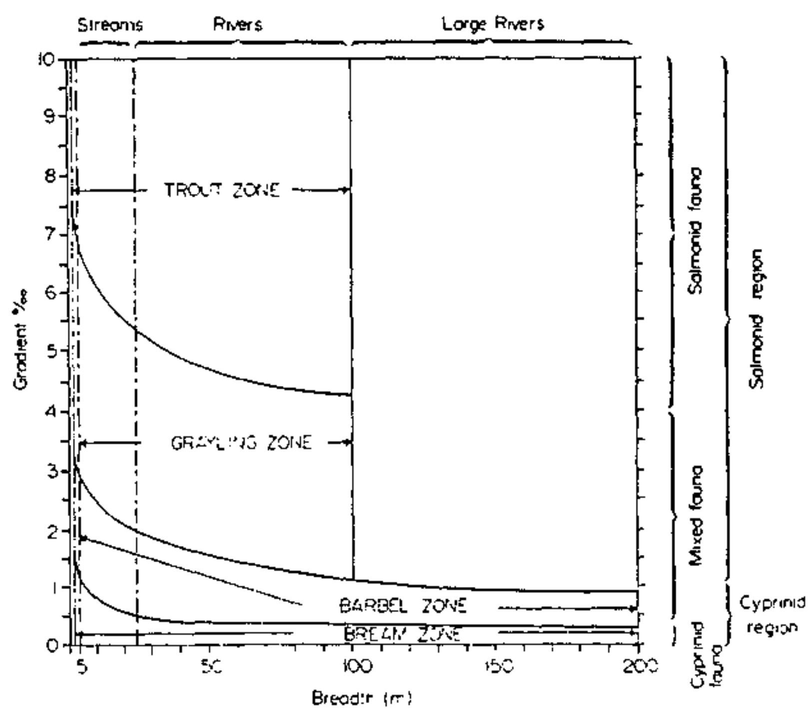


Figure 1. Slope graph, showing the relationship between gradient, river width and fish faunal zone (after Huet 1959).

Fish zone Characteristic fish	TROUT (*Salmo trutta*)	GRAYLING (*Thymallus thymallus*) (MINNOW)* (*Phoxinus phoxinus*)	BARBEL (*Barbus barbus*) (CHUB)* (*Leuciscus cephalus*)	BREAM (*Abramis brama*)
Fish fauna	Salmonids[1]	Mixed fauna Salmonids dominant	Mixed fauna Cyprinids dominant	Cyprinid fauna with predators
Principal fish in order of abundance				
Dominant and sub-dominant	Trout (*Salmo trutta*) Salmon Parr* (*Salmo salar*)	Grayling (Minnow)* Species found in trout zone	Rheophilic Cyprinids[2]	Limnophilic Cyprinids[5] Associated Cyprinids[3] Associated predators[4]
Common	Bullhead (*Cottus gobio*) Minnow (*Phoxinus phoxinus*)	Rheophilic Cyprinids[2]	Associated Cyprinids[3] Associated predators[4]	— —
Rare		Associated Cyprinids[3] Associated predators[4]	Species fauna in trout zone Limnophilic Cyprinids[5]	Rheophilic Cyprinids

(1) Salmonids—Brown trout (*Salmo trutta*), Grayling (*Thymallus thymallus*), Salmon* (*Salmo salar*).
(2) Rheophilic Cyprinids—Barbel (*Barbus barbus*), Chub (*Leuciscus cephalus*), Hotu (*Chondrostoma nasus*), Gudgeon (*Gobio gobio*)*.
(3) Associated Cyprinids—Roach (*Rutilus rutilus*), Rudd (*Scardinius erythrophthalmus*), Dace (*Leuciscus leuciscus*).
(4) Associated predators—Pike (*Esox lucius*), Perch (*Perca fluviatilis*), Eel (*Anguilla anguilla*).
(5) Limnophilic Cyprinids—Carp (*Cyprinus carpio*), Tench (*Tinca tinca*), Bream (*Abramis brama*).

Table 4. Occurrence and relative abundance of principal fish species in each Huet fish faunal zone (after Hawkes 1975).

Other systems have attempted to relate river zones to stream order (eg Kuehne 1962; Stauffer et al 1975), but all such systems have failed to demonstrate that there is an ecological justification for treating river systems as a series of discrete species assemblages with distinct faunal breaks (Hawkes 1975, Matthews 1986). In reality, there is likely to be either a zonation in which optimal conditions inhabited by one community fade into and overlap with those of the next (Balon and Stewart 1983), forming broad transition zones, or a continuum of community change regulated by the interaction of abiotic and biotic factors (Zalewski and Naiman 1985). However, zonation schemes have been and continue to be a useful first approximation for the purposes of water management.

In addition to changes in fish assemblage, fish production is likely to exhibit an increase downstream, due largely to the longitudinal changes in nutrient status and temperature. However, whether or not the potential for production is actually realised depends largely upon the constraints of the physical habitat, ie the carrying capacity.

Other environmental variables are generally either strongly intercorrelated with those mentioned above, or are more randomly distributed variables (often due to anthropogenic habitat modification) which serve to alter habitat size and diversity, eg depth, vegetation cover. It could be said that the former type of variable largely dictates the potential species assemblage (based on physico-chemical tolerance, ie autecological variables), whilst the latter type of variable dictates the degree of synecological interaction (and thus population size and the degree of competitive exclusion - see subsection on synecological effects).

More recent studies have taken both types of variable and have looked in detail at individual species requirements, assuming no empirical species associations. The most notable examples are the Habitat Suitability Index (HSI) (US Fish & Wildlife Service 1980) and HABSCORE (Milner et al 1985). In this study it is not possible to look at environmental variables in this detail because:

i) Such data are not available (see Section 3.1.3);

ii) the habitat requirements of indigenous species in the UK have not been given rigorous and quantitative attention, such that it would not be possible to discern whether the absence or reduced abundance of a species were a habitat effect or a water quality effect. However, much of the variation in physical habitat can be obviated using Huet's classification, by simple measurement of slope and channel width (see Section 3.3.4), and the degree of synecological interaction can be estimated by using qualitative assessments of habitat size and diversity as surrogates where possible.

In addition to autecological factors operating within catchments, the limits to geographical range must be considered when interpreting observations on species assemblage. For instance, within the UK the bullhead is generally found only in England and Wales, being rare in Scotland (Smyly 1957).

Synecological effects

Where two species occur within their autecological range, their co-existence will depend upon the degree of overlap between their respective niches. If their niches are identical, the competitive exclusion principle states that co-existence is not possible. Such a drastic result is not usual in natural and diverse habitats, since similar species invariably differ in their requirements or behaviour, or else adapt in the face of strong competition. Stochastic variation also limits the amount of niche overlap (Henderson 1985). However, competitive and predator-prey interactions will influence the relative and absolute abundance of both species and populations to a greater or lesser extent.

As habitat diversity is reduced, the scope for differences in niche declines, such that the influence of competitive interaction on species abundance is enhanced and species exclusion becomes more likely.

Post-impact studies on channelised river stretches have shown reductions in fish species diversity and abundance (eg Portt *et al* 1986, Swales 1982). It is evident that in this way modification of the physical habitat can bring about similar responses in the fish community as would be expected from deterioration in water quality.

Stochastic effects

Variability in species assemblage and population size may result from random fluctuation of environmental conditions. The relative strengths of year classes within unimpacted populations give an indication of this. Newman and Waters (1989) in a 3 year study of the brown trout production of contiguous sections of an entire stream found varying densities and production of fish both between years and river sections. Relative differences between sections were nearly constant and were attributed to habitat differences, but overall differences between years were attributed to variability in recruitment.

Stocking activities

Fish stocking greatly obscures the natural distribution of fish species (Wheeler 1974) and their abundance. Through stocking it is possible to maintain a population where self-maintenance by reproduction is not possible due to physico-chemical constraints. However, if there are no barriers to their movement it has been shown that stocked fish rapidly migrate away from the site of introduction (Linfield 1985), presumably to areas with preferred habitat and water quality characteristics. Such post-stocking movements may therefore serve to offset the confounding effects of stocking, since stocked populations will tend to redistribute based on the prevailing environmental conditions.

Watersheds produce a barrier to fish movement which for many species can only be realistically overcome through human intervention by stocking. This can produce a patchy geographical distribution which bears no relationship to physical or chemical constraints. Examples of such distributions are the barbel and grayling.

Since stocking activities generally only affect species of importance to anglers, the distribution of minor species should not be obscured. However the use of some minor species as livebait may result in some transfer of fish.

Acclimation to toxic stress

Acclimation to chronic sub-lethal concentrations of certain pollutants can occur, thereby reducing the sensitivity to toxic stress. This may occur at the individual level, in the form of physiological or biochemical adaptation. An example of this is the induction of metallothioneins to counteract the effects of high heavy metal concentrations (Kay et al 1987). Long-term exposure in relation to population turnover (eg metalliferous leachates from old mine tailings) may excert a selective pressure leading to genetic change.

Acclimation is likely to produce a range of sensitivities to a given toxicant which will tend to mask any relationship between fishery status and toxicant concentration.

3.3 DATA ANALYSIS

3.3.1 Introduction

A number of approaches were used in the data analysis, including standard univariate and multivariate techniques, but also a rule-based approach, using an 'expert system', and the toxicity-based approach discussed in Section 2.2.3.

Expert systems seek to define sets of rules in a non-linear manner which best describe different aspects of a database. This approach was used in this study to investigate the environmental requirements for the presence of selected species.

A computer program was constructed based on the toxicity-based studies outlined in Section 2.2.3 which sums the toxicity exerted by a number of toxicants. Each toxicant concentration was converted to a fraction of the relevant toxicity standard, and these fractions were than summed. This approach assumes that the effects of the toxicants concerned are simply additive, which is not necessarily the case. It also assumes that all toxicants exerting a significant stress are included in the calculation. The toxicity standards used in the program were:

i) 48 hour LC50 for rainbow trout;

ii) 48 hour LC50 for roach;

iii) threshold (no-effect) LC50 for roach.

The toxicological database was more complete for rainbow trout than for roach, with the result that the variation in roach toxicity with ancillary variables such as hardness, temperature and pH was not included in the TOXIC program (see Appendix D). Although individual species have a different sensitivity to each toxicant, rainbow trout and roach were considered to be indicative of the sensitivity of salmonids and cyprinids respectively.

The parameters included in the program were NH_3N, DO, NO_2N, Cd, Cr, Cu, Ni, Pb, Zn, CN, and phenol. The variation in toxicity of certain parameters with prevailing water quality (eg hardness, DO, free CO_2) was accounted for as far as toxicological knowledge allowed. Details of the calculation of toxicity for each toxicant are given in Appendix D. The program was tested on historical data from the River Trent, previously analysed using the toxicity-based approach to predict the presence/absence of fisheries (Alabaster et al 1972). In testing the TOXIC program, toxic scores were related to a subjective assessment of fishery class (Appendix E).

For the purposes of this study, CN and phenol were considered irrelevant since no measure of them was recorded in this database. Of the other toxicants, NH_3N and DO were consistently recorded. NO_2N was recorded at about half of the sites and metals were seldomly recorded. Two approaches to toxicity summation were made:

i) NH_3N, DO and NO_2N only were summed;

ii) all toxicants were summed, assuming that if a toxicant was not measured its toxic effect was negligible.

The above approaches were used on the whole database, on single catchments and on habitat-based groups of sites. The advantage of using single catchments is that species are less likely to be patchily distributed due to problems of colonisation, since watersheds are the largest barrier to fish movement. Grouping sites based on their habitat attributes seeks to minimise the variability introduced by habitat, as discussed in Section 3.2.2, and thus minimise its confounding influence on the detection of water quality effects.

It was also envisaged that temporal trends would be investigated in this study, but no sites within the database had sufficient information to attempt this.

'Less than' values were taken as the Limit of Detection (ie $<x = x$) in the analysis, except where the TOXIC program was used and showed that a 'less than' value had a significant toxicity, in which case the value was taken as zero. 'Less than' values having a significant toxic effect were used if actual readings (ie not 'less than values) existed to support the use of the the values as the Limit of Detection. When ancillary variables were not available to calculate the toxicity of parameters included in the TOXIC program, regression equations derived from the database were used to estimate values.

Finally, it should be emphasised that the sites included in this database were not selected within a meaningful statistical framework, such that inferences from the data analysis cannot justifiably be used outside of the confines of the database.

3.3.2 Analysis of the undivided database

Preliminary investigation of variation in the water quality variables suggested that there were better correlations between fishery parameters and the arithmetic means of the water quality variables than with other measures of central tendency, such as the median, geometric mean, or mode, or with percentiles.

The numbers of sites falling in each class of the NWC classification are shown below:

CLASS	1A	1B	2	3	4
NO OF SITES	10	36	30	23	1

(sites outside of England and Wales are not included here)

Fisheries

Fisheries parameters are summarised in Table 5.

The total biomass of fish varied from zero to 11,280 g 100 m^{-2}, with a mean biomass of 1,871 g 100 m^{-2}. The distribution of these biomass estimates was highly skewed, however, with 37% of the records less than 1000 g 100 m^{-2}, and only 6% of the records greater than 5000 g 100 m^{-2}. The median biomass was 1457 g 100 m^{-2}, and half of all the records had a biomass between 521 and 2700 g 100 m^{-2}.

The total density of fish was recorded for only 94 of the 112 samples, and varied from zero to 253.2 fish 100 m^{-2}. Again, the distribution of these density estimates was highly skewed, with 63% of the records showing less than 25 fish 100 m^{-2}, and only 15% of the records showing more than 100 fish 100 m^{-2}. The median density was 13.45 fish 100 m^{-2}, with half of the records having densities between 5.87 and 52.30 fish 100 m^{-2}.

Table 5 - Summary statistics for fishery parameters

Variable	Minimum	Mean	Maximum	Standard deviation
Total biomass (g 100 m^{-2})	0	1871	11280	1894
Total density (no 100 m^{-2})	0	41.6	253.2	5.87
Number of species present	0	6.6	13	3.1
Biomass of salmonids (g 100 m^{-2})	0	289.3	2864	578.1
Density of salmonids (no 100 m^{-2})	0	13.8	191.8	33.9
Biomass of coarse fish (g 100 m^{-2})	0	1312.0	10730	1876.3
Density of coarse fish (no 100 m^{-2})	0	16.2	128.3	22.2
Biomass of eels (g 100 m^{-2})	0	278.4	2744	481.1
Density of eels (no 100 m^{-2})	0	9.57	200.7	28.9

Variable	No. of samples	1st quartile	Median	3rd quartile
Total biomass (g 100 m^{-2})	112	521	1457.5	2700.5
Total density (no 100 m^{-2})	94	5.87	13.45	52.30
Number of species present	107	4	6	9
Biomass of salmonids (g 100 m^{-2})	112	0	0	312
Density of salmonids (no 100 m^{-2})	107	0	0	6.1
Biomass of coarse fish (g 100 m^{-2})	111	51.3	724	1872
Density of coarse fish (no 100 m^{-2})	94	1.7	7.5	18.7
Biomass of eels (g 100 m^{-2})	111	0	88	329.5
Density of eels (no 100 m^{-2})	93	0	0.8	3.7

The number of species was recorded for 107 of the 112 samples, and varied from zero to 13, with a mean of 6.6. The distribution of the number of species was slightly less skewed than for total biomass or for total density, with 13% of the records having 3 or less species, and 20% having 10 or more species. The median number of species was 6, and half of the records had between 4 and 9 species represented. Only 5 of the sites had no fish at all.

The biomass of salmonids for which records were available ranged from zero to 2,864 g 100 m^{-2}, with a mean biomass of 289.3 g 100 m^{-2}. However, 69 of these sites recorded no salmonids, and the mean biomass of sites with salmonids was 753 g 100 m^{-2}. Similarly, although the recorded density of salmonids ranged from zero to 191.8 100 m^{-2}, with a mean density of 13.8 100 m^{-2}, 71 of the 107 records of salmonid density were zero. The mean density of salmonids at sites with salmonids was 41 fish 100 m^{-2}.

The average biomass of coarse fish (all fish excluding salmonids and eels) was 1312 g 100 m^{-2}, with a maximum of 10,730 g 100 m^{-2}. The distribution of biomass was, however, highly positively skewed, and 45% of the records had a biomass of less than 1000 g 100 m^{-2}. Only 16% of the records had a biomass greater than 3000 g 100 m^{-2}. The average density of coarse fish (excluding eels) was 16.2 fish 100 m^{-2}, with a maximum of 128.3 fish 100 m^{-2}. Again, the distribution of densities was highly positively skewed, with 57% of the sites recording less than 10 fish 100 m^{-2}. Only 21% of the sites recorded more than 20 fish 100 m^{-2}.

The average biomass of eels was 278.4 g 100 m^{-2}, with a maximum of 2744 g 100 m^{-2}. However, 41 sites had no eels, and the mean biomass of the 70 sites with eels was 441.5 g 100 m^{-2}. Similarly, while the average density of eels was 9.57 fish 100 m^{-2}, with a maximum of 200.7 fish 100 m^{-2}, the mean density of the sites with eels was 14.1 fish 100 m^{-2}.

NWC class

The average values of fishery and invertebrate parameters in each NWC class are given in Table 6 and Figures 2, 3 and 4. Differences between classes were tested for significance using a two-sample t-test. Figure 5 shows the variation in NWC class with physical habitat, as defined by Huet's fish zones (discussed at length in Sections 3.2.2 and 3.3.4), and helps to explain the differences observed in Table 6. It is evident that within the database class 1A sites are restricted to the trout and grayling zones, ie the upper fast-flowing sections of rivers dominated by salmonids, and that the grayling, barbel and bream zones have on average much poorer water quality than the trout zone.

There were no significant differences in mean total biomass between classes 1A, 1B, 2, and 3. Mean total density was significantly ($p<0.01$) lower in classes 1B, 2, and 3 than in class 1A, due mainly to the greater emphasis placed on the capture of small individuals in salmonid dominated areas (all class 1A sites being in the trout or upper grayling zone), as discussed in Section 3.2.1.

Classes 1B and 2 had significantly ($p<0.05$ & $p<0.01$ respectively) more species than class 1A, and class 2 had significantly more species than class 3 ($p<0.01$). The low species number in class 1A is due to the small number of species able to withstand the physical nature of the trout zone (see Section 3.2.2), whereas the low species number in class 3 is due to water quality constraints, since class 3 sites in this study are generally further downstream where the potential number of species is greater. Species number is maximal in this database in class 2, presumably where sites are sufficiently downstream to allow a high number of potential species, and where water is of sufficient quality to allow at least most of that potential to be realised. The single recorded location in class 4 had no fish of any kind, which is again probably due to water quality constraints.

Table 6 - Fishery and invertebrate status by NWC class

(Mean values with standard error below each mean. Biomass values in $g 100m^{-2}$, density values in no $100m^{-2}$)

NWC	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP	ASPT
1A	2322	135.3	4.5	1620	76.6	661	40	40.7	17.9	109	6.02
	345	28.4	0.4	155	18.7	270	16	19.8	7.8	17	0.13
1B	1700	22.2	6.9	236	3.2	237	1227	2.3	16.6	91	4.50
	350	4.7	0.4	58	1.1	67	335	0.9	4.0	8	0.31
2	2470	16.5	8.9	3	0	103	2365	0.4	16.2	81	4.22
	397	5.9	0.5	2	0	35	384	0.2	5.8	7	0.18
3	1546	15.7	5.5	17	0	247	1283	1.7	14.1	63	3.83
	339	4.1	0.8	13		90	308	0.5	4.0	10	0.19

tBIO = total biomass.
coaBIO = coarse fish biomass
tDENS = total density
coaDENS = coarse fish density
NofSP = number of species
ASPT = Average Score Per Taxon

SalBIO = Salmonid biomass.
eelBIO = eel biomass
SalDENS = salmonid density
eelDENS = eel density
BMWP = Biological Monitoring Working Party Score

t-Test results:

tBIO

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

tDENS

	1B	2	3
1A	**	**	**
1B		ns	ns
2			ns

NofSP

	1B	2	3
1A	*	**	ns
1B		**	ns
2			**

salBIO

	1B	2	3
1A	**	**	**
1B		**	**
2			ns

salDENS

	1B	2	3
1A	**	**	**
1B		*	*
2			ns

eelBIO

	1B	2	3
1A	*	**	ns
1B		ns	ns
2			ns

coa BIO

	1B	2	3
1A	ns	**	*
1B		*	ns
2			*

eelDENS

	1B	2	3
1A	**	**	*
1B		ns	ns
2			*

coaDENS

	1B	2	3
1A	ns	ns	ns
IB		ns	ns
2			ns

BMWP

	1B	2	3
1A	ns	ns	ns
1B		ns	*
2			ns

ASPT

	1B	2	3
1A	*	**	**
1B		ns	ns
2			ns

* $p<0.05$
** $p<0.01$
ns not significant

Figure 2. Mean fish biomass by NWC class.

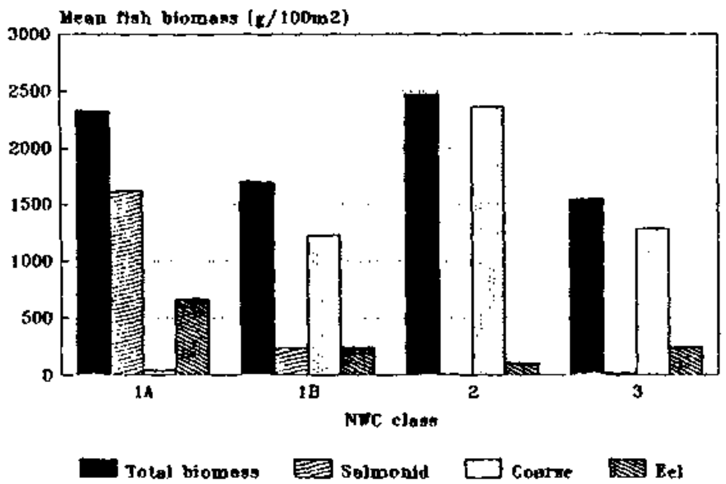


Figure 3. Mean fish density by NWC class.

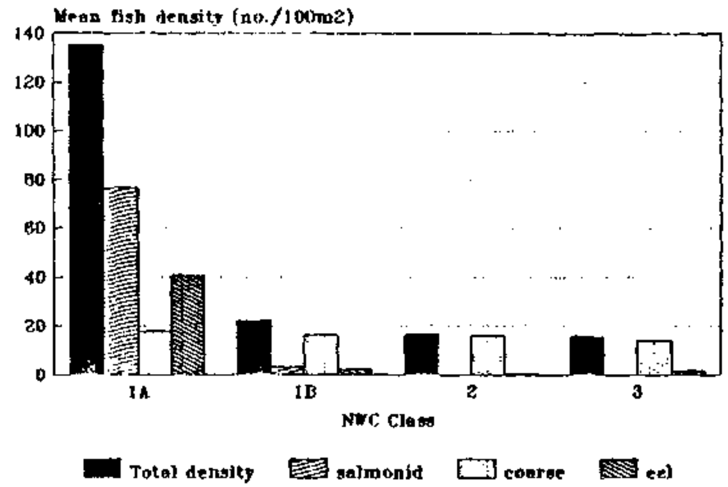


Figure 4. Mean number of fish species and invertebrate scores by NWC class.

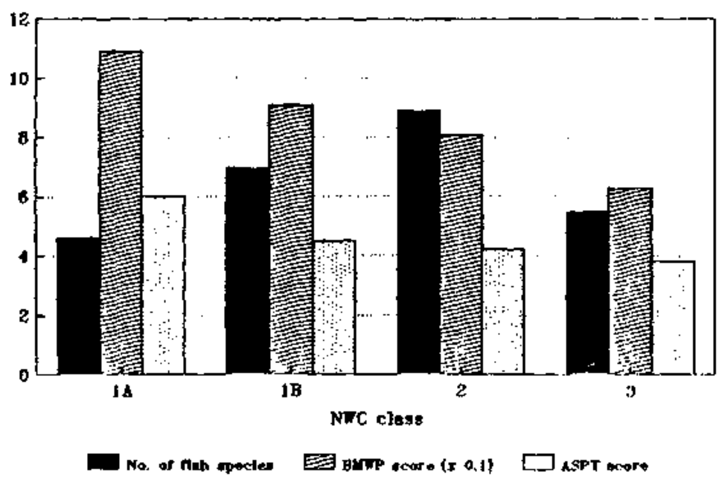
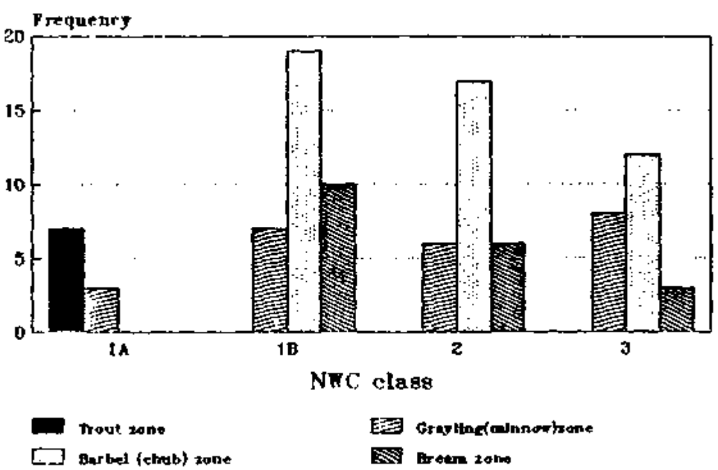


Figure 5. NWC class frequency distribution, by fish zone.



Class 1A had significantly higher salmonid ($p<0.01$) biomass than all other classes, generally higher eel biomass, but a much lower biomass of coarse fish. It is tempting to ascribe the decline in salmonid biomass with NWC class to water quality effects; however, it is evident from Figure 5 that all class 1A sites lay in salmonid-dominated habitats (trout and grayling zones), whereas only a relatively small proportion of sites in the other NWC classes are naturally salmonid-dominated. Class 2 had the largest mean biomass of coarse fish, almost no salmonids and a low mean biomass of eels. This coincides with the highest mean number of species, observed earlier, and is again likely to occur at sites with a reasonable compromise between autecological constraints and water quality constraints.

Figure 3, showing mean fish densities by NWC class, is greatly influenced by sampling methodology differences which obscure any possible water quality effects.

Regarding invertebrate status, both BMWP score and ASPT scores declined with NWC class. Since both scores produce higher values in gravelly riffle zones and all class 1A sites are located in the fast flowing trout and upper grayling zones, it is not surprising that class 1A had the highest values of both (although only ASPT is significantly different from the other classes). The differences in mean ASPT between classes 1B, 2 and 3 are not significant, but the significant difference in BMWP score between classes 1B and 3 ($p<0.05$) may be a real water quality effect.

Habitat

The four variables used to measure the physical parameters of the sample sites are summarized in Table 7.

The mean width of the sampled rivers ranged from 1.25 m to 30 m, with a mean of 8.23 m. Only 7% of the sites had widths greater than 15 m. Depths were measured on only about half of the sites and varied from 17 cm to 150 cm, with a mean of 83.7 cm. The altitude of the sites varied

Table 7 - Summary of physical parameters

Variable	No of samples	Minimum	Mean	Maximum	Standard deviation
Mean width (metres)	110	1.25	8.23	30.0	4.69
Mean depth (cm)	60	17	83.7	150	34.4
Altitude (metres)	112	7	56.2	226	44.0
Slope (%)	112	0.02	0.33	3.00	0.49

from 7 m to 226 m, with a mean of 56.2 m, but only 11% of these sites were at altitudes greater than 100 m. The slope of the rivers at the sampled sites varied from 0.02% to 3.00%, with a mean of 0.33%, but only 6% of these sites had slopes greater than 1.00.

<u>Water quality</u>

Water quality variables sufficiently complete to be worth including in the analysis are summarized in Table 8.

Mean pH, available for all sites, varied from 6.49 to 8.24, with a mean of 7.73 and a standard deviation of 0.35. This range is not extreme enough to produce a toxic effect, unless variability about the mean is high. The UK EQS for pH is that 95% of samples should lie within the range 6 - 9. Only 10 (9%) of the sites had a mean pH less than 7.25. Half of the sites had a mean pH between 7.55 and 7.96.

Average temperature, available for all sites, varied between 7.4°C and 15.2°C, with a mean of 10.6°C and a standard deviation of 1.4°C. Half of the average temperatures were between 9.6°C and 11.7°C.

Table 8 - Summary of water quality parameters

Variable	No of samples	Minimum	Mean	Maximum	Standard deviation
pH	112	6.49	7.73	8.24	0.35
Average temperature (°C)	112	7.4	10.6	15.2	1.4
Suspended solids ($mg\ l^{-1}$)	89	2	21	110	18.9
BOD ($mg\ l^{-1}$)	109	1.22	3.24	18.66	2.20
Dissolved oxygen ($mg\ l^{-1}$)	111	6.14	10.26	12.99	1.09
NH_3N ($mg\ l^{-1}$)	111	0.006	0.608	9.153	1.051
NO_2N ($mg\ l^{-1}$)	45	0.01	0.17	1.15	0.20
NO_3N ($mg\ l^{-1}$)	51	0.09	5.53	18.93	4.29
PO_4 ($mg\ l^{-1}$)	60	0.01	1.34	10.16	2.02
Average hardness ($mg\ l^{-1}$ $CaCO_3$)	46	7	244	482	143
Alkalinity ($mg\ l^{-1}$ $CaCO_3$)	62	5	166	530	106
Conductivity ($\mu s\ cm^{-1}$)	85	50.4	694.5	1715	427.5

Average suspended solids, available for only 89 of the 112 sites, varied from 2 $mg\ l^{-1}$ to 110 $mg\ l^{-1}$ with a mean of 21 $mg\ l^{-1}$ and a standard deviation of 18.9 $mg\ l^{-1}$. The distribution of values was positively skewed, with a median of 15.6 $mg\ l^{-1}$ and only 12 (13%) of the values greater than 32.9 $mg\ l^{-1}$. This parameter may well produce impacts on fish populations in this study, since Alabaster and Lloyd (1982) conclude that an average concentration of 25 $mg\ l^{-1}$ may be sufficient to impair fish yield.

Average values of BOD, available for only 109 of the 112 sites, varied from 1.22 $mg\ l^{-1}$ to 18.66 $mg\ l^{-1}$ with a mean of 3.24 $mg\ l^{-1}$ and a standard deviation of 2.20 $mg\ l^{-1}$, but only 6 of the sites had a BOD greater than 6.2 mg l^{-1}. The median BOD was 1.73 mg l^{-1} and the first and third quartiles were 2.2 mg l^{-1} and 3.6 mg l^{-1} respectively. This parameter, though not itself toxic, may act as a surrogate for sediment quality, since the settlement of particulates with a high BOD is likely to lead to reduced interstitial dissolved oxygen levels. This may then have an impact on those fish that spawn in or on the bottom substrate, those fish with a benthic habit, or the benthic invertebrate food of such fish.

Average values of dissolved oxygen, available for all but one of the 112 sites, varied from 6.14 mg l^{-1} to 12.99 mg l^{-1} with a mean of 10.26 mg l^{-1} and a standard deviation of 1.09 mg l^{-1}. The mandatory 50% compliance values for EC designated salmonid and cyprinid waters are 9 and 7 mg l^{-1} respectively (see Appendix A.1). A high variability about the lower mean values in the database could cause an impact upon fish.

Average values of total ammoniacal nitrogen (NH_3N), available for all but one of the 112 sites, varied from 0.006 mg l^{-1} to 9.153 mg l^{-1} with a mean of 0.608 mg l^{-1} and a standard deviation of 1.051 mg l^{-1}. This range is sufficient to cause toxic effects on fish populations, although the toxicity will vary not only with total ammonia concentration but also with the distribution between ionised and un-ionised forms. This distribution varies with pH, temperature, alkalinity, conductivity, and DO. The UK mandatory EQS for EC designated salmonid and cyprinid waters is an annual average of 0.78 mg l^{-1}, although it is proposed that this is changed to a 95%ile value (see Appendix A.1). Again, the distribution of pH values was positively skewed, with a median of 0.303 mg l^{-1} and first and third quartiles of 0.111 mg l^{-1} and 0.608 mg l^{-1} respectively. Only 15 of the sites had NH_3N values greater than 1.313 mg l^{-1}.

The remaining measures of water quality were available for no more than half of the sites.

Average values of nitrite (NO_2N), available for only 45 of the sites, varied from 0.01 mg l^{-1} to 1.15 mg l^{-1} with a mean of 0.17 mg l^{-1} and a standard deviation of 0.20 mg l^{-1}. The distribution of NO_2N values was again positively skewed, with a median of 0.12 mg l^{-1} and first and third quartiles of 0.05 mg l^{-1} and 0.19 mg l^{-1} respectively. Only 11 of the sites had NO_2N values greater than 0.20 mg l^{-1}. At these mean concentrations, NO_2N is not likely to influence fish populations.

In addition to low sampling frequencies, the following variables are not actually toxic to fish at environmental concentrations, but may either influence fish populations by affecting productivity (NO_3N and PO_4) or may act as rough indicators of productivity (hardness, alkalinity and conductivity).

Average values of nitrate (NO_3N), available for 51 of the sites, varied from 0.09 mg l^{-1} to 18.93 mg l^{-1} with a mean of 5.53 mg l^{-1} and a standard deviation of 4.29 mg l^{-1}. Only 9 (18%) of the values were greater than 9.50 mg l^{-1}.

Average values of orthophosphate (PO_4), available for 60 of the sites, varied from 0.01 mg l^{-1} to 10.16 mg l^{-1} with a mean of 1.34 mg l^{-1} and a standard deviation of 2.02 mg l^{-1}. The distribution of PO_4 values was positively skewed, with a median of 0.55 mg l^{-1} amd first and third quartiles of 0.08 mg l^{-1} and 1.49 mg l^{-1} respectively. Only 3 (6%) of the PO_4 values were greater than 5.09 mg l^{-1}.

Average hardness, measured as mg l^{-1} of $CaCO_3$, and available for only 46 of the sites, varied from 7 mg l^{-1} to 482 mg l^{-1} with a median of 244 mg l^{-1} and a standard deviation of 143 mg l^{-1}. 20 (44%) of the sites had average hardness values less than 245 mg l^{-1}, the remaining 26 (56%) having average hardness values greater than 245 mg l^{-1}.

Average alkalinity, measured at pH 4.5 in mg l^{-1} $CaCO_3$, and available for 62 of the sites, varied from 5 mg l^{-1} to 530 mg l^{-1} with a mean of 166 mg l^{-1} and a standard deviation of 106 mg l^{-1}.

Mean conductivity, measured in µs cm^{-1} at 25 °C, and available for only 85 of the 112 sites, varied from 50 to 1715 µs cm^{-1}, with a mean of

694.5 µs cm^{-1} and a standard deviation of 427.5 µs cm^{-1}. Half of the mean conductivities were between 386 and 924 µs cm^{-1}.

Correlations

In the discussion of correlations which follows, it must be stressed that each of the correlation coefficients is based on a different number of observations, depending on the completeness of the data matrix. As an aid to their interpretation, therefore, the correlations which are larger in absolute value than the tabulated values for the appropriate number of degrees of freedom at the 0.05, 0.01, and 0.001 levels of probability are indicated. Because simultaneous testing of large numbers of correlation coefficients by taking them two at a time overestimates the number of significant correlations, these indications should be treated with some caution.

Correlations between fishery status, invertebrate status and habitat variables are given in Table 9.

Total biomass was negatively correlated with mean channel width (MWIDTH). This may be due to a decrease in the ratio of bankside cover to open water with increasing channel width, but may just reflect the difficulty of quantitatively fishing deeper waters. Total density was negatively correlated with channel width and mean depth (MDEPTH), and positively correlated with slope (SLOPE). This is consistent with variations in fish sampling methodologies mentioned earlier and probably cannot be attributed to variation in habitat. The number of species was positively correlated with depth of the river, but negatively correlated with the altitude (ALT) and slope, which would be expected from known autecological ranges (see Section 3.2.2).

Salmonid biomass and density were both positively correlated with altitude and slope, but salmonid density alone was negatively correlated with mean depth. Sampling artefacts affected salmonid density, but all of these correlations would nevertheless be expected.

Table 9 - Correlations between fishery status, invertebrate status and habitat variables

(Values are Pearson coefficients of correlation)

	MWIDTH	MDEPTH	ALT	SLOPE	BMWP	ASPT
tBIO	$-.20^{1}$	.12	-.08	.01	.09	-.01
tDENS	$-.21^{1}$	$-.64^{3}$	.18	$.66^{3}$	.19	.43
NOofSP	.04	$.35^{1}$	$-.29^{2}$	$-.31^{2}$	.03	-.08
salBIO	-.08	-.25	$.36^{3}$	$.65^{3}$	$.30^{1}$	$.59^{2}$
salDENS	-.14	$-.59^{3}$	$.48^{3}$	$.86^{3}$	.23	$.58^{2}$
eelBIO	-.12	$-.41^{2}$	.02	.19	-.03	.27
coaBIO	-.14	$.30^{1}$	-.19	$-.24^{1}$	.00	$-.43^{1}$
eelDENS	-.01	$-.47^{3}$	.02	$.23^{1}$	.00	.22
coaDENS	$-.37^{3}$	-.25	$-.29^{2}$	.09	.16	-.11
BMWP	.13	.00	-.01	.13		$.61^{3}$
ASPT	.03	-.50	$.55^{2}$	$.63^{3}$		

^{1} $p<0.05$
^{2} $p<0.01$
^{3} $p<0.001$

The biomass of coarse fish was positively correlated with depth and negatively correlated with slope, while in contrast, the density of coarse fish was negatively correlated with both width and altitude. Again, density values were affected by sampling artefacts, but coarse fish biomass behaved according to autecological constraints and restrictions imposed by living space. Both eel biomass and eel density were negatively correlated with mean depth, but eel density was positively correlated with slope.

Regarding invertebrate status, BMWP score and ASPT were positively correlated with each other ($p<0.001$) as might be expected, and were also correlated with altitude and slope ($p<0.01$ and 0.001 respectively). The latter correlations are likely to be partly a product of poorer water quality at downstream locations (with respect to the study sites in this database), but mainly reflect a general tendency for upstream, high gradient sites to have gravelly and well-oxygenated substrates which support good populations of high-scoring invertebrates. Armitage et al (1983) found a similar variation in ASPT in a survey of unpolluted sites, generally declining from upland to lowland areas. There was no significant correlation between total biomass and BMWP score or ASPT. The positive correlations between both salmonid biomass and density and ASPT ($p<0.01$), and negative correlation between coarse fish biomass and ASPT ($p<0.05$), were similarly mainly due to intercorrelations with physical habitat variables.

Correlations between habitat and water quality variables are given in Table 10.

Table 10 - Correlations between physical and water quality variables

Variables	Width	Depth	Alt	Slope
pH	.100	-.148	-.308[3]	-.467[2]
COND	-.148	.396[2]	-.419[3]	-.477[3]
TEMP	-.019	.048	-.337[3]	-.032
SSLDS	.083	.024	-.165	-.301[2]
BOD	-.094	.184	-.285[2]	-.192
DO	.001	-.126	.179	.125
NH_3N	-.161	.088	-.041	-.075
NO_2N	.020	.142	-.209	-.315[1]
NO_3N	.047	.683[1]	-.467[3]	-.402[2]
PO_4	-.079	.274	-.319[1]	-.304[1]
THARD	.201	.312	-.221	-.662[3]
ALKY	.206	.189	-.141	-.532[3]

Channel width was not appreciably correlated with any of the water quality variables, but river depth was correlated with COND and NO_3N. Altitude was negatively correlated with pH, conductivity (COND), temperature (TEMP), BOD, NO_3N and PO_4. Slope was negatively correlated with pH, COND, SSLDS, NO_2N, NO_3N, PO_4, total hardness (THARD) and alkalinity (ALKY).

Correlations between fishery status and water quality variables with sufficient data are given in Table 11.

Table 11 - Correlations between fishery status and water quality variables

	tBIO	tDENS	NofSP	salB10	CoaB10	EelB10	SalDENS	CoaDENS	EelDENS
pH	.05	-.30[2]	.19	-.29[2]	.16	-.04	-.37[3]	-.19	-.03
COND	.08	-.53[3]	.39[3]	-.47[3]	.29[2]	-.25[1]	-.51[3]	-.19	-.31[2]
TEMP	.10	.14	-.04	-.06	.10	.05	-.01	.10	.18
SSLDS	-.21[1]	-.15	-.11	-.22[1]	-.19	.19	-.23[1]	-.17	.10
BOD	-.15	-.24[1]	-.11	-.31[3]	-.02	-.15	-.29[2]	.01	-.15
DO	.11	.07	.25[2]	.21[1]	.03	.10	.13	-.07	.04
NH_3N	-.22[1]	-.29[2]	-.33[3]	-.21[1]	-.13	-.12	-.30[1]	-.12	-.14
NO_2N	-.25	-.54[2]	-.15	-.38[1]	-.07	-.29	-.44[2]	-.27	-.27
NO_3N	-.03	-.22	.30[1]	-.28[1]	.14	-.26	-.30[1]	.10	-.20
PO_4	-.11	-.45[3]	.13	-.37[2]	.05	-.19	-.41[2]	-.15	-.25
THARD	-.02	-.71[3]	.21	-.65[3]	.24	-.30[1]	-.60[3]	-.41[2]	-.39[2]
ALKY	-.11	-.55[3]	-.01	-.35[2]	.04	-.25[1]	-.43[3]	-.35[1]	-.32[1]

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

Total biomass was negatively correlated with SSLDS and NH_3N. As stated previously, both parameters are likely to have an ecotoxicological effect at the concentrations recorded in the database.

Total density was negatively correlated with pH, COND, BOD, NH_3N, NO_2N, PO_4, THARD and ALKY. Density values were affected by sampling artefacts that will correlate with any parameter that has a longitudinal trend. Most of these parameters have a tendency to increase downstream (see Table 10), and therefore negatively correlate with density, however, total NH_3N does not correlate with depth, altitude or slope and this correlation may therefore be indicative of a toxic effect.

Number of species (NofSP) was positively correlated with COND, dissolved oxygen (DO) and NO_3N, and negatively correlated with NH_3N. The correlations with DO and mean NH_3N may be due to toxic effect, but are likely to be confounded by the strong influence of habitat on NofSP.

Both salmonid biomass and density were negatively correlated with pH, COND, SSLDS, BOD, NH_3N, NO_3N, PO_4, THARD and ALKY. Most of these parameters show longitudinal changes down rivers and are therefore likely to be acting as surrogates for changes in physical habitat (see Table 10), from upstream salmonid habitats to lowland cyprinid habitats. However, ammonia and suspended solids may be contributing to the change in dominance from salmonids to cyprinids. Salmonid biomass alone was positively correlated with DO. Coarse fish biomass was positively correlated with COND, which is acting as a surrogate for decreasing slope and altitude (see table 10), and thus increasing habitat suitability for cyprinids. Eel biomass and density and coarse fish density were negatively correlated with COND, THARD and ALKY.

Toxicity analysis

TOX1 values were usually identical to TOX3 values, due to a lack of metals data in the database. Exceptions to this included some sites in the NRA South West and Severn Trent regions where TOX1 scores were sometimes significantly higher than TOX3 scores, due to the the presence of metals.

An explanation of the various toxicity scores used is given in Table 12.

Table 13 shows correlations of fishery parameters with the mean toxicity scores at each site, as calculated by the TOXIC program (see Section 3.3.1 and Appendix D).

Since toxicity score based on roach are not relevant to salmonids, these correlations have been omitted from the database. However, since there is more account taken of the variation in toxicity with fluctuating ancillary variables (eg alkalinity) in the calculation of trout toxicity scores, correlations between these scores and coarse fish parameters have been included.

Table 12 - Explanation of toxicity score codes

CODE	TOXICITY SCORE
TOXT1A	Mean proportion of the 48h LC50 concentration to rainbow trout, using all toxicants (ie including metals).
TOXT3A	As above, but using only NH_3N, DO and NO_2N.
TOXR1A	Mean proportion of the 48h LC50 concentration to roach, using all toxicants.
TOXR3A	As above, but using only NH_3N, DO and NO_2N.
TOXR1T	Mean proportion of the threshold LC50 concentration to roach, using all toxicants.
TOXR3T	As above, but using only NH_3N, DO and NO_2N.

Table 13 - Correlations between fishery status, invertebrate status and toxicity scores

(Values are Pearson coefficients of correlation)

	tBIO	tDENS	NOofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
TOXT1A	-.25[1]	.22	-.49[3]	.20	.43[3]	.00	-.31[2]	.19	-.15
TOXT3A	-.32[2]	-.25[1]	-.48[3]	-.25[1]	-.20[1]	-.18	-.21	-.14	-.14
TOXR1A	-.25[1]	-.17	-.42[3]			-.09	-.20	-.01	-.21
TOXR3A	-.25[1]	-.24[1]	-.40[3]			-.12	-.17	-.16	-.19
TOXR1T	-.28[1]	-.15	-.42[3]			-.11	-.23[1]	.01	-.23
TOXR3T	-.28[2]	-.25[1]	-.44[3]			-.14	-.19	-.16	-.19

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

NofSP was highly negatively correlated ($p<0.001$) with all six toxicity scores, which is a surprising result considering the strong habitat influence on this parameter. Apart from strong correlations ($p<0.001$) between TOXT1A and both altitude and slope, there was no intercorrelation between toxicity scores and habitat. This is to be expected since the most influential factor upon each toxicity score is NH_3N, which was not correlated with any habitat variable (see Table 10). It would seem therefore, that the correlations between NoFSP and toxicity score (except for TOXT1A) may be indicative of real toxic effects. Mean toxicity score values have the following ranges within the database.

TOXT1A	0-0.24 (24%)	TOXR1A	0-0.43 (43%)	TOXR1T	0-0.44 (44%)
TOXT3A	0-0.15 (15%)	TOXR3A	0-0.43 (43%)	TOXR3T	0-0.44 (44%)

Since a mean value of 0.27 - 0.39 was found to indicate fishless sites in the Trent (see Appendix E), the values in the database were expected to indicate significant effects on fish.

Total fish biomass was negatively correlated with all toxicity scores, but at lower levels of significance than NoFSP. Total density was negatively correlated with all three TOX3 scores, but not TOX1 scores.

Salmonid biomass was negatively correlated ($p<0.05$) with TOXT3A (summing NH_3N, NO_2N and DO toxicity only), and salmonid density was negatively correlated with TOXT3A ($p<0.05$) but highly positively correlated with TOXT1A ($p<0.001$). The latter correlation was due to intercorrelation of TOXT1A with altitude and slope. Coarse fish biomass was negatively correlated with TOXR1T ($p<0.05$) and TOXT1A ($p<0.01$), but again the latter correlation was due to intercorrelation with habitat variables.

A rough indication of the completeness of the water quality data in this study was gained by reference to fishless sites within the database. Mean TOXT1A scores for the fishless sites AN9, NW2 and NW3 were 8, 8 and 3% respectively. These values are very low when compared with the threshold mean value for fish absence from historical Trent data of 27% (see Appendix E), which suggests that other toxicants which have not been monitored are having an effect. This can only be expected with the paucity of data evident within the database.

The corresponding TOXR1A values for the same three fishless sites were 43, 3 and 10% respectively. The high value at site AN9 was due to NH_3N, and highlights the danger of not taking into account the variation in factors which affect NH_3N toxicity (the roach toxicity scores assume constant values of alkalinity and DO). For this reason, the roach toxicity scores should be treated with some caution, and they have not been used in the multiple regression analysis that follows.

Multiple Regression Analysis (MRA)

In order to assess the relative importance of measured environmental variables to fishery status, stepwise MRA was performed, using those variables for which a possible causative influence was detected in the correlation analysis. For only some of these variables were there sufficient data to warrant their inclusion as explanatory, or regressor, variables in such regressions, namely width, altitude, and slope of the river at the point of sampling, and average pH, TEMP, BOD, DO and NH_3N. Attempts to include the variables depth, COND, SSLDS, NO_2N, NO_3N, PO_4, THARD and ALKY in the regression reduced the available data to a small subset of the sites, and are not, therefore, reported here.

While it is tempting to try to interpret the regression coefficients directly, it is important to remember that there are marked intercorrelations between the explanatory variables which make such interpretation unreliable, and in extreme cases may ascribe the wrong sign to the coefficient. The relative contributions to the explained variance are given for each variable only if the degree of intercorrelation between regressor variables was relatively low, and even these should be treated with caution. The principal benefit of the regressions is in assessing how much of the variability of the dependent variable can be accounted for by the explanatory variables.

Density parameters were omitted from the analysis since they were so influenced by sampling artefacts. Results of the analysis are given in Table 14.

Table 14 - Stepwise multiple regression analysis of the undivided database

(Values are R-squared values, indicating the percentage of total variance accounted for)

	NofSP		NofSP
	0%		0%
+ SLOPE	9.7%	+ TOXT3A	23.1%
+ NH_3N	22.8%	+ SLOPE	33.6%
+ ALTITUDE	25.9%	(ALTITUDE)	
(DO)			
	tBIOMASS		**tBIOMASS**
	0%		0%
+ NH_3N	5.6%	+ TOXT3A	9.6%
+ mWIDTH	12.0%	+ mWIDTH	16.7%
	salBIO		**salBIO**
	0%		0%
+ SLOPE	48.0%	+ SLOPE	47.8%
+ NH_3N	50.6%	+ TOXT3A	51.7%

(Variables in parentheses were included in the analysis but not used by the program)

Slope, NH_3N and altitude explained 25.9% of the variance in NofSP, with NH_3N accounting for about half of this. Mean DO did not account for any variance. When NH_3N and DO were replaced by TOXT3A the explained variance increased to 33.6%.

NH_3N and mean channel width explained 12% of the variance in total fish biomass. When NH_3N was replaced with TOXT3A this figure increased to 16.7%.

Slope and NH_3N accounted for 50.6% of the variance in salmonid biomass and, as would be expected, slope was responsible for almost all of this. The explained variance only increased to 51.7% when NH_3N was replaced by TOXT3A.

No MRA was performed on eel biomass since there was no indication of toxic effect in the univariate correlation analysis.

Principal Components Analysis

Principal components analysis was performed on both the fisheries and environmental (water quality and habitat) data, in order to identify independent sources of variation within the database and identify site groupings on the basis of that variation. However, no clear interpretation of this analysis was possible, and results are therefore not presented in this report.

Association analysis

Sites were clustered on the basis of the similarity between their fish species assemblages, in an attempt to identify discrepancies brought about by impoverished water quality. However, the resultant site groupings did not allow a clear distinction to be made between habitat and water quality effects. It is clear from this analysis that either habitat variables need to be held constant (ie all sites must come from a similar habitat type) in order to identify water quality effects, or water quality needs to be held constant and in a pristine state in order to assess the habitat-based pattern of species assemblage, over which the effect of water quality might then be discerned.

Rule-based discrimination using the expert system approach

The following sets of rules were produced by a computer-based expert system which attempted to predict the presence/absence of selected species using the available physico-chemical data.

Rivers with no salmonids:

a) If BOD > 3.3 mg l^{-1} or SLOPE $\leq 0.185\%$
and $NH_3N > 0.097$ mg l^{-1}
and ALT < 90 m
and DO < 10.97 mg l^{-1} and pH > 7.56
Then river will contain no salmonids (93%)

b) If BOD > 3.3 mg l^{-1} or SLOPE < 0.18%
and NH_3N > 0.097 mg l^{-1}
and ALT < 90 m
Then river will contain no salmonids (67%)

c) If NH_3N > 0.097 mg l^{-1}
and ALT < 90 m
and DO < 10.97 mg l^{-1} and pH > 7.56
Then river will contain no salmonids (75%)
Else river will contain salmonids (97%)

In each set of rules a mean total NH_3N concentration of greater than 0.1 mg l^{-1} was linked to an absence of salmonids. The importance of altitude and slope is confirmed as is a dependence on high DO concentrations (a mean of greater than 11 mg l^{-1}). Healthy salmonid populations should exist at lower mean DO levels than this, and it would be interesting to see if this value was reduced by the introduction of new sites into the database. The pH requirement of less than 7.56 for the presence of salmonids is clearly a function of their predominance in upland areas. The rules concerning BOD may be linked to spawning bed quality but are more likely to be due to intercorrelation with other variables.

Presence or absence of brown trout:

a) If BOD < 2.0 mg l^{-1}
and SLOPE > 0.17% or pH < 7.1
and DO > 9.88 mg l^{-1}
Then river contains brown trout (100%)

b) If SLOPE > 0.17% or pH < 7.13
and DO > 9.88 mg l^{-1}
Then river contains brown trout (61%)
Else river does not contain brown trout (87%)

Similar comments to those above apply here. The DO requirement was lower, at 9.9 mg l^{-1}, but trout should thrive at DO levels lower than this providing fluctuations about the mean are not severe. Neither set of rules included a threshold for NH_3N.

Presence or absence of roach:

If SLOPE < 0.28%
and BOD > 1.9 mg l^{-1}
and ALT < 83 m and MWIDTH > 4.0 m
Then river will contain roach (77%)
Else river will not contain roach (84%)

Roach were generally not found at slopes of greater than 0.28%, altitudes greater than 83 m and in rivers less than 4 m wide. However, no consideration was made of water quality requirements, and this may reflect a lack of sensitivity to poor water quality.

Presence or absence of bullheads:

If pH < 7.44 and NH_3N < 0.307 mg l^{-1}
and TEMP < 12.8 or SLOPE > 0.21%
and BOD < 1.7 mg l^{-1} or MWIDTH < 7.2 m
Then river contains bullheads (82%)
Else river does not contain bullheads (88%)

Bullheads were found at higher levels of NH_3N than salmonids, at up to 0.31 mg l^{-1}. They would appear to be generally restricted (at least within this dataset) to rivers less than 7.2 m wide and with slopes greater than 0.21%. This may be a function of their preference for shallow water, or may be due to a lack of recording of such minor species in cyprinid fisheries. Similarly the temperature thresholds of 12.9°C and pH threshold of 7.44 are also likely to be sampling artefacts.

The reported sensitivity of this species to suspended solids (Section 2.2.2) is not evident from the expert system analysis of this dataset.

3.3.3 Single catchment studies

Nene catchment (NRA Anglian Region)

The database holds 13 sites from the Nene catchment, 7 of which are located in the Willow Brook. Altitude ranged from 18 to 89 m, thus obviating much of the longitudinal variation in habitat evident when analysing the database intact. Although zinc pollution is known to have been a historical problem in the Willow Brook (Solbe 1973), no zinc data was available for the period under study. NWC class ranged from 2 to 3.

Table 15 shows correlations between those variables with sufficient data; a coefficient of >0.602 is required for statistical significance at the 95% level, providing no observations are missing.

Table 15 - Correlations between fishery status, invertebrate status and physico-chemical variables in the Nene catchment

(Values are Pearson coefficients of correlation)

	tBIO	NOofSP	salBIO	eelBIO	coaBIO	BMWP	ASPT
MWIDTH	-.13	.40	-.25	-.09	-.12	.44	.17
ALT	-.20	-.59	-.30	-.11	-.19	-.90[2]	-.73
SLOPE	-.03	-.48	.07	.07	-.06	-.56	-.21
pH	.11	.20	.41	.41	-.02	.47	.71
TEMP	-.29	-.35	-.15	-.09	-.32	-.28	-.21
SSLDS	.14	-.17	-.24	.02	.18	-.22	-.43
BOD	.08	-.43	-.11	.15	.06	-.07	-.04
DO	.53	.66[1]	.64[1]	.27	.53	.41	.46
NH_3N	-.30	-.62[1]	-.17	-.14	-.31	-.43	-.34
NH_3N*	-.42	-.85[3]	-.32	-.22	-.42	-.43	-.34
NO_2N	-.37	-.47	-.36	-.29	-.34	-.06	-.28
NO_3N	.00	.45	-.15	-.08	.03	.18	.06
ALKY	.66	.49	.48	.36	.66	.31	.26
BMWP	-.14	.25	.51	.14	-.24		.90
ASPT	.09	.25	.75	.40	-.06		

NH_3N* = excluding the extreme value at site AN11.
[1] = $p<0.05$
[2] = $p<0.01$
[3] = $p<0.001$

There was a significant ($p<0.01$) negative correlation between altitude and BMWP score, due largely to the uppermost sites (in Willow Brook) being the most polluted. This was not reflected by the correlation between total fish biomass and altitude ($p>0.05$), but the correlation between the NofSP and altitude approaches significance at the 95% level. Regarding specific relationships with water quality, a highly significant correlation was evident between NoFSP and NH_3N ($p<0.001$). This correlation was improved greatly by excluding the extreme NH_3N value of 9.153 mg l^{-1} at site AN11 (see Figure 6). NoFSP also showed a significant positive correlation with dissolved oxygen (DO) ($p<0.05$), although this is likely to be due to intercorrelation with NH_3N since the lowest mean DO for any site was only 8.24 ± 1.71 mg l^{-1}, unlikely to cause stress. Total fish biomass, coarse fish biomass, BMWP score and ASPT showed similar trends with NH_3N and DO, but not at any high level of statistical significance.

Such observations suggest that, at least in lowland areas where the potential number of fish species is high, species assemblage may be a better indicator of water quality than population size, since the latter takes no account of the pollution sensitivity of the species constituting the population.

No relationship was evident between fishery status and invertebrate status. The relationship between salmonid biomass and ASPT is nearly significant at the 95% level, but only 2 sites had any salmonids present.

Table 16 gives correlations between fishery parameters and toxicity scores.

The number of fish species was negatively correlated with all 6 toxicity scores, as might be expected from its correlation with NH_3N. No other correlations were significant at the 95% confidence level.

Figure 6. The relationship between mean total ammonia and the number of fish species in the Nene catchment.

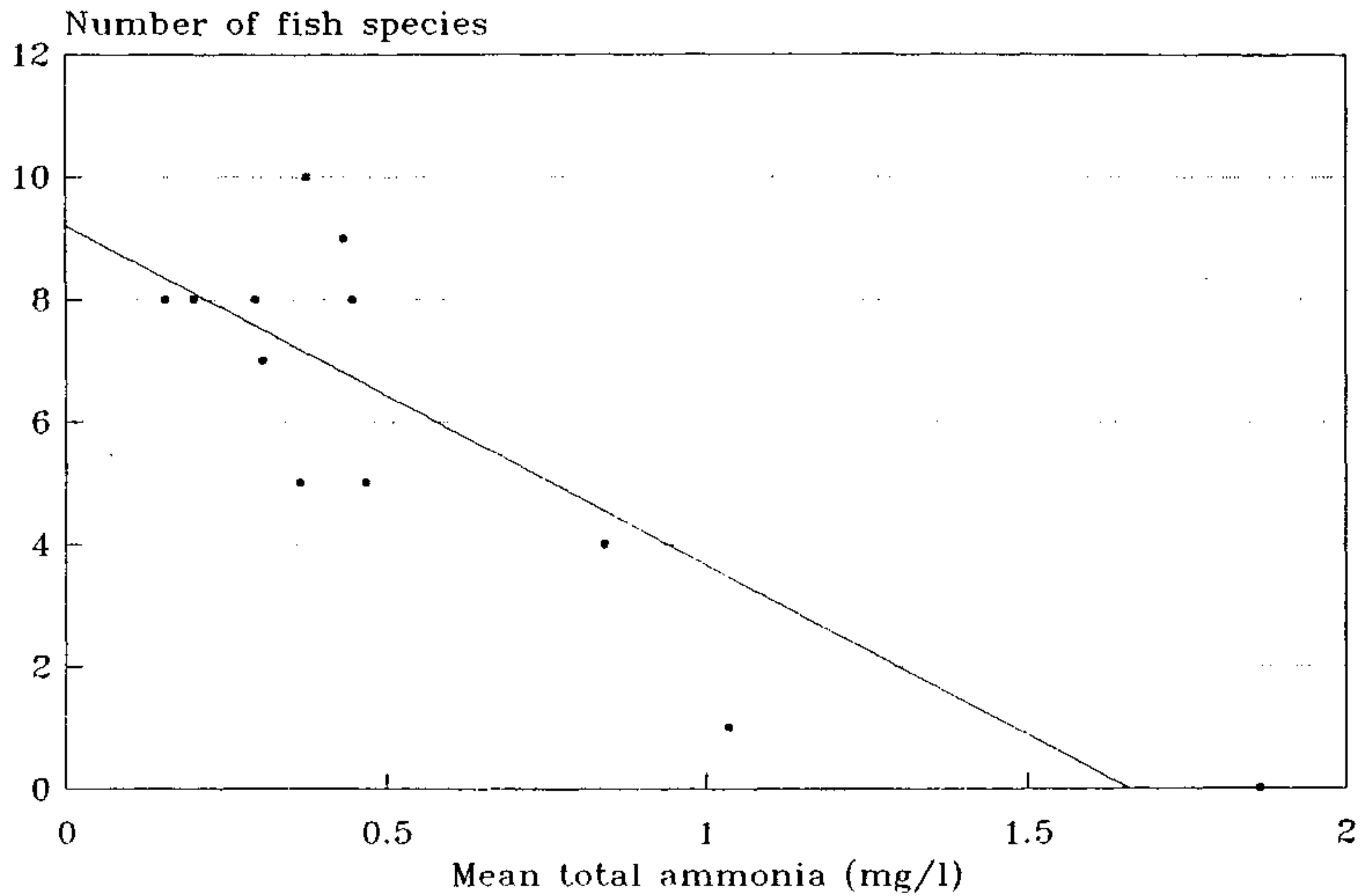


Table 16 - Correlations between fishery status and toxicity scores in the Nene catchment

(Values are Pearson coefficients of correlation)

	tBIO	NofSP	salBIO	eelBIO	coaBIO
TOXT1A	-.48	-.81[3]	-.26	-.24	-.49
TOXT3A	-.48	-.81[3]	-.26	-.24	-.49
TOXR1A	-.38	-.79[2]		-.17	-.40
TOXR3A	-.38	-.79[2]		-.17	-.40
TOXR1T	-.43	-.82[3]		-.19	-.45
TOXR3T	-.43	-.82[3]		-.19	-.45

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

The range in toxicity scores is given below:

TOXT1A	2 - 15	TOXR1A	1 - 43	TOXR1T	2 - 44
TOXT3A	2 - 13	TOXR3A	1 - 43	TOXR3T	2 - 44

As is evident from these ranges, all measured toxicity was due to NH_3N, NO_2N and DO. Maximum values are high enough to cause toxic effects.

Table 17 shows the results of MRA.

NH_3N accounted for 72.2% of the variance in the number of species. The intercorrelation between total NH_3N and DO is clear from the variance accounted for by DO alone and and that accounted for by NH_3N, DO and NO_2N combined (84.2%). When the latter 3 variables are toxicity-weighted and combined into TOXT3A the explained variance decreases to 65.6%. A much lower proportion of the variance in total biomass and coarse fish biomass (essentially the same values since little salmonid biomass is present in the catchment) was explained by these variables.

Table 17 - Multiple regression analysis of sites in the Nene catchment

(Values are R-squared values, indicating the percentage of total variance accounted for)

	NofSP	tBIOMASS	coaBIOMASS
NH_3N:	72.2%	17.3%	17.5%
DO:	46.0%	28.2%	27.6%
TOXT3A:	65.6%	22.6%	23.6%
NH_3N, DO, NO_2N:	84.2%		

Perry catchment (NRA Severn Trent region)

The database holds 6 sites from the Perry catchment, all of which have repeat datasets. Each dataset was treated as a separate site for the purpose of the correlation analysis. All sites were of class 1B. Altitude ranged from 60 to 80 m, and slope from 0.04 to 0.13%, allowing little longitudinal habitat variation.

Table 18 shows correlations between variables with sufficient data. A coefficient of 0.632 was required for significance at the 95% level, provided no observations were missing. Significant negative correlations between salmonid biomass and slope ($p<0.001$), salmonid biomass and altitude ($p<0.05$), and salmonid density and slope ($p<0.05$) are the reverse of natural trends (salmonids might be expected to favour the headwaters of lowland rivers) and are likely to be intercorrelations with true influencing variables.

A negative trend (verging on significance at $p<0.05$) between salmonid biomass and NH_3N was also evident, which may be affecting salmonid distribution. The highest mean NH_3N value was 0.674 mg l^{-1}, compared to the mandatory standard of 0.78 mg l^{-1} as a 95%ile for waters designated under the EC Freshwater Fish Directive (see Appendix A.1). Moreover, all fishery statistics (except eel biomass and density) show a negative trend with NH_3N, all verging on significance at the 95% level. The partitioning of total ammonia between the ionised and un-ionised fractions is not known.

Table 18 - Correlations between fishery status, invertebrate status and physico-chemical variables in the Perry catchment

(Values are Pearson coefficients of correlation)

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP
MWIDTH	-.52	-.42	-.23	-.17	-.34	-.44	-.37	-.50	-.34	-.17
MDEPTH	.45	.40	.19	.17	.31	.38	.30	.48	.32	.17
ALT	-.43	-.55	-.15	-.71[1]	-.53	-.02	-.14	-.07	-.43	-.79
SLOPE	-.52	-.70[1]	-.47	-.89[3]	-.69[1]	.41	-.32	.41	-.62	-.80
pH	.23	.46	.69[1]	.55	.42	-.69[1]	.26	-.66[1]	.56	.57
COND	-.04	.02	-.33	-.05	-.05	.58	-.24	.61	-.03	.64
TEMP	.07	.27	.34	.47	.36	-.52	.04	-.55	.19	.32
SSLDS	-.26	-.13	-.38	-.17	-.13	.40	-.37	.41	-.19	.73
BOD	.28	-.18	-.24	-.32	-.23	.26	.40	.25	-.11	-.68
DO	-.06	-.03	.38	-.06	-.08	-.39	.11	-.32	.09	-.26
NH_3N	-.55	-.51	-.59	-.60	-.45	.57	-.57	.54	-.58	-.05
THARD	-.28	-.20	-.68[1]	-.24	-.17	.50	-.40	.50	-.30	.06
ALKY	.58	.75[2]	.30	.83[2]	.71[1]	-.34	.40	-.38	.75[1]	.15
BMWP	-.07	.12	-.04	.52	.11	.40	-.39	.57	.08	

[1] = $p<0.05$
[2] = $p<0.01$
[3] = $p<0.001$

The positive correlation between pH and NofSP is likely to be due to intercorrelation, since the lowest mean pH recorded in the catchment was 7.63 ± 0.19 and therefore non-toxic (the UK standard is that 95% of samples should lie within the range pH 6 to 9 - see Appendix A.1).

BMWP score was uncorrelated with both total fish biomass and NofSP, but showed quite strong positive trends with salmonid and eel biomass.

Correlations between fishery parameters and toxicity scores are given in Table 19.

The spuriously high correlation coefficients with roach toxicity scores are due to all of these scores having identical and very low values. Trout toxicity scores were very low, and no significant correlations were evident between them and fishery parameters.

Table 19 - Correlations between fishery status, invertebrate status and toxicity scores in the Perry catchment

(Values are Pearson coefficients of correlation)

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
TOXT1A	-.16	.10	-.13	.22	.15	.23	-.39	.25	-.07
TOXT3A	-.16	.10	-.13	.22	.15	.23	-.39	.25	-.07
TOXR1A	1.00	1.00	1.00			1.00	1.00	1.00	1.00
TOXR3A	1.00	1.00	1.00			1.00	1.00	1.00	1.00
TOXR1T	1.00	1.00	1.00			1.00	1.00	1.00	1.00
TOXR3T	1.00	1.00	1.00			1.00	1.00	1.00	1.00

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

Table 20 gives the results of MRA.

Table 20 - Multiple regression analyses of sites in the Perry catchment

(Values are R-squared values, indicating the percentage of total variance accounted for)

	NofSP	tBIOMASS	salBIOMASS	coaBIOMASS
NH_3N:	34.3%	30.5%	36.1%	32.2%
SLOPE:	ns	27.2%	79.8%	

Over 30% of the variance in the number of species, total biomass, salmonid biomass and coarse fish biomass was explained by NH_3N. However, no toxic effect due to NH_3N was indicated by the toxicity analysis, suggesting that: either recorded NH_3N concentrations are not reflecting its true impact (perhaps due to the effect of infrequent episodic events); or that NH_3N is acting as a marker for another variable. Slope accounted for 79.8% of the variance in salmonid biomass, but the relationship is negative, ie the reverse of the expected habitat effect. Furthermore, the range in slope was narrow, only 0.04 to 0.13%, and was not thought to have much influence. Slope accounted for 27.2% of the variance in total biomass, but is not likely to be a causative factor. The graph of salmonid biomass against NH_3N is given in Figure 7.

Tavy (NRA South West Region)

Seven sites in the Tavy catchment are included in the database, all being of class 1A and ranging in altitude from 12 to 225 m. Table 21 shows correlations for variables with sufficient data. Due to the small number of sites, a coefficient of 0.878 was required for significance at the 95% level (providing there were no missing data).

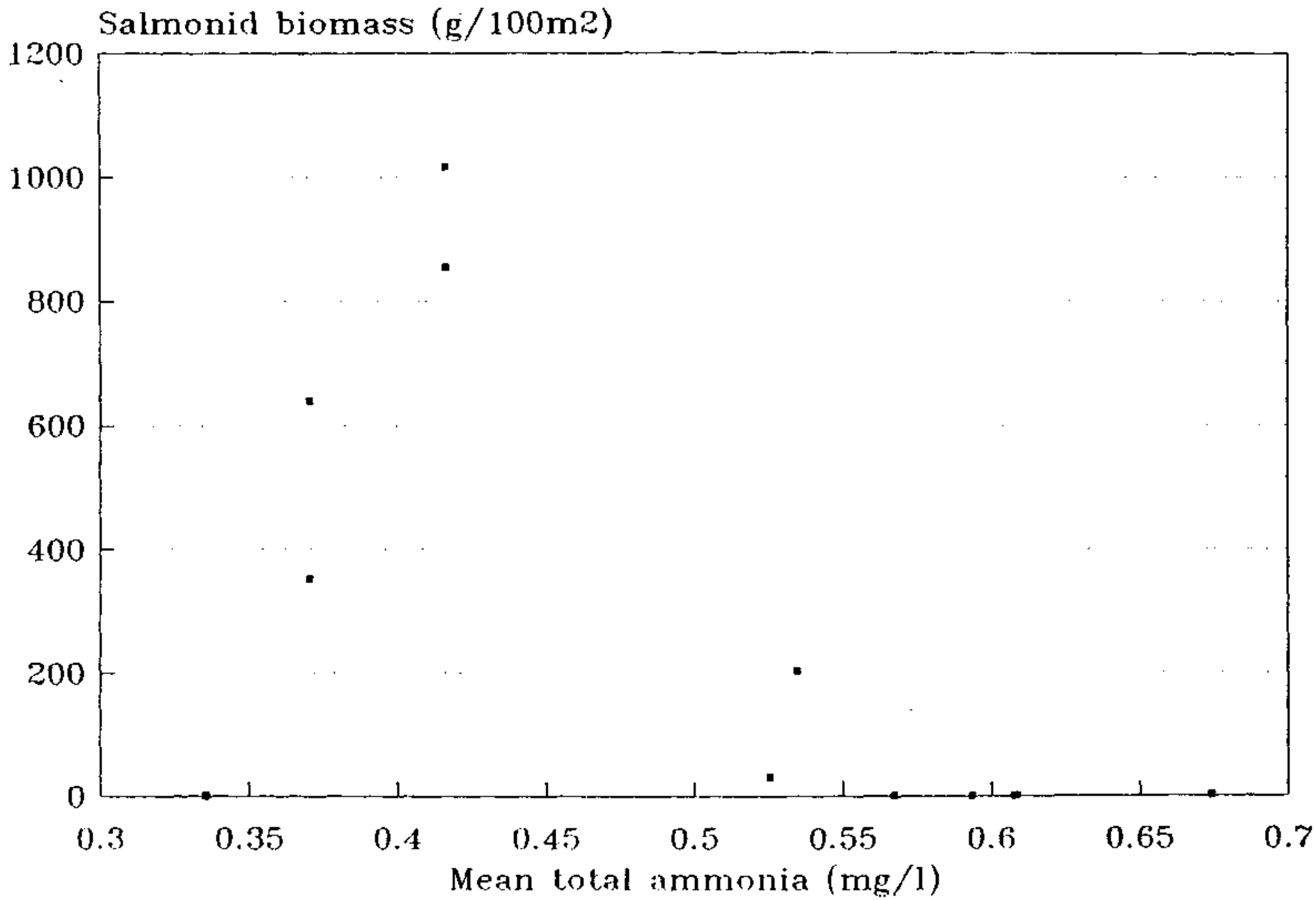
Table 21 - Correlations between fishery status and physico-chemical variables in the Tavy catchment

(Numbers are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
MWIDTH	.10	-.23	-.06	-.39	-.14	.39	-.87	.28	-.86
ALT	-.68	-.70	-.64	-.61	.46	-.49	-.66	-.61	-.68
SLOPE	-.54	-.24	-.79	-.02	.86	-.66	-.10	-.76	-.17
pH	.79	.56	.65	.76	-.63	.54	.77	.61	.72
COND	.56	.55	.69	.63	-.49	.32	.89[1]	.43	.89[1]
TEMP	.85	.60	.70	.70	-.73	.65	.66	.74	.65
SSLDS	.66	.65	.80	.48	-.61	.53	.70	.66	.72
BOD	.54	.14	.18	-.01	-.53	.70	-.47	.69	-.47
DO	-.24	-.19	-.54	-.43	.50	-.03	-.82	-.16	-.87
NH_3N	.84	.63	.66	.21	-.66	.94[1]	.01	.99[2]	-.02
NO_2N	.79	.65	.71	.15	-.68	.91[1]	.07	.99[2]	.06
NO_3N	.53	.56	.60	.65	-.47	.27	.90[1]	.41	.92
PO_4	.73	.67	.49	.41	-.62	.68	.35	.83	.37
ALKY	.54	.51	.74	.58	-.48	.31	.87	.40	.86

[1] = $p<0.05$
[2] = $p<0.01$
[3] = $p<0.001$

Figure 7. Salmonid biomass vs mean total ammonia in the Perry catchment.



Strong positive correlations were evident between eel density/biomass and both mean total NH_3N and NO_2N. In contrast, there were negative associations (though not significant at the 95% level) between salmonid density and both NH_3N and NO_2N. These latter observations are unlikely to be indications of toxic effect (unless the monitoring regime was not reflecting true environmental levels), since recorded NH_3N and NO_2N concentrations in the catchment were low (the highest mean concentrations being 0.13 ± 0.07 mg l^{-1} and 0.03 ± 0.03 mg l^{-1} respectively, at Langham Wood, approximately 1 km downstream from a sewage treatment works discharge and a waste disposal tip). It is probable that the above relationships are due to a general downstream increase in productivity, owing to an increase in nutrient status, which coincides with a reduction in habitat favourable to young salmonids with increasing distance downstream. This is indicated by correlations between coarse fish biomass and both NO_3N concentration ($p<0.05$) and mean conductivity ($p<0.05$), and between salmonid density and slope.

All TOX3 (NH_3N, DO and NO_2N combined) scores were calculated as being zero in the toxicity analysis. Some metals data were available in the Tavy catchment such that mean TOX1 scores ranged from 0 to 8%, with the exception of one site, SW3, which produced a mean TOXT1A of 24%; this site had the second lowest salmonid biomass of the 7 sites in the catchment (1185 g 100 m^{-2}). TOX1 scores could only be calculated for 4 sites, so no correlation matrix is given.

3.3.4 Habitat-based division of the database

Introduction

Sites were grouped on the basis of Huet's slope rule (see Figure 1 and Section 3.2.2), which uses slope and channel width as habitat descriptors. The classification consists of 4 fish zones:

"The Trout zone" - this is the zone of steepest gradient and consequently highest current velocity. Waters are always cool and well oxygenated, usually headwaters, and the fish fauna is dominated by the brown trout.

"The Grayling (minnow) zone" - this consists of quite rapidly flowing water with a substrate of finer material than the trout zone. It supports a mixed fauna of salmonids and running water cyprinids (see Table 4).

"The Barbel (chub) zone" - current velocity is moderate, quiet water is much more frequent than in the Grayling zone. The fish fauna is dominated by running water cyprinids, with sluggish-water cyprinids ('accompanying' cyprinids) and still-water cyprinids inhabiting the quieter zones. Summer temperatures are moderately high.

"The Bream zone" - this is usually the lower stretch of rivers, with low current velocities, high summer temperatures and fairly low dissolved oxygen levels. The fish fauna is dominated by sluggish-water and still-water cyprinids.

From Table 4 it is evident that there is a great deal of overlap in species assemblage between zones. The zones are often grouped into:

i) "the Salmonid region", comprising the trout and grayling zones;

ii) "the Cyprinid region", comprising the barbel and bream zones.

<u>Variations in fishery status between zones</u>

Table 22 and Figures 8, 9 and 10 show the basic differences in fish populations between fish zones. All differences were tested for significance using a two-sample t-test.

Figure 8 shows salmonid biomass declining from the Trout zone through to the lower zones, with the mean biomass of 1513 g 100 m^{-2} in the trout zone being significantly higher than all other zones, and the mean biomass of 346 g 100 m^{-2} in the grayling zone being significantly higher than that in the barbel zone (37 g 100 m^{-2}). In contrast, mean coarse fish biomass increased from trout zone (64.5 g 100 m^{-2}) to bream zone (2717 g 100 m^{-2}), all differences being significant except that between

Figure 8. Mean fish biomass by fish zone

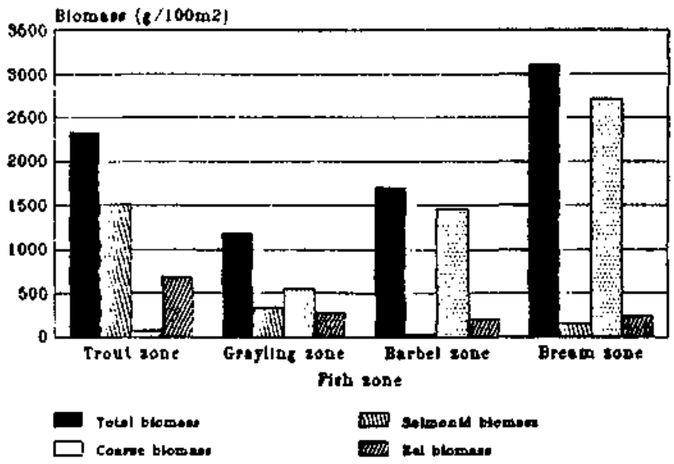


Figure 9. Mean fish density by fish zone

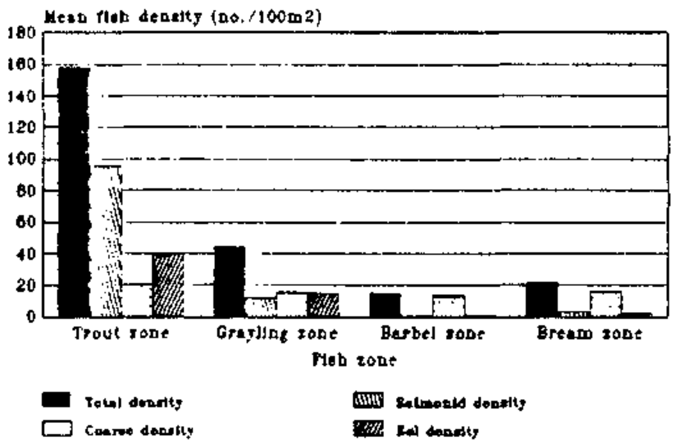
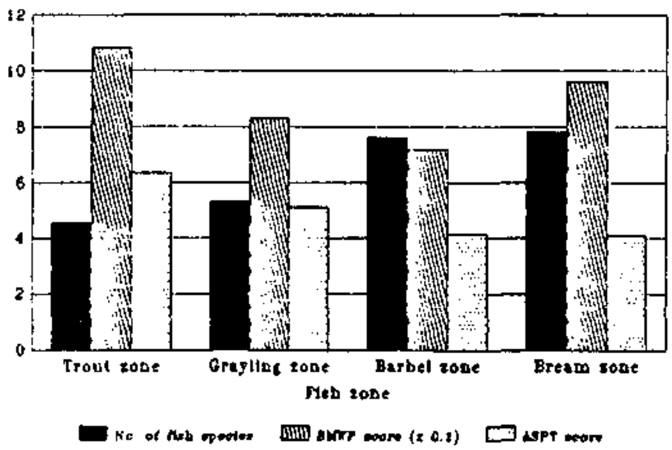


Figure 10. Mean number of fish species and invertebrate scores by fish zone.



trout and grayling zones. Mean eel biomass showed a decline from the trout zone to the lower zones, but only the difference between trout zone (685 g 100 m^{-2}) and barbel zone (207 g 100 m^{-2}) is significant.

Table 22 - Fishery and invertebrate status by fish zone

(Mean values with standard error given below each mean. Biomass values in g 100m^{-2}, density values in No. 100 m^{-2}).

ZONE	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP	ASPT
TROUT	2327	157.8	4.5	1513	95.5	685	64.5	39.8	20.1	108	6.34
	382	21.7	0.5	232	15.5	238	21.0	17.8	7.1	10	0.61
GRAYLING	1184	43.7	5.3	346	11.7	276	554	14.7	15.1	83	5.12
	269	10.7	0.6	93	3.3	99	220	7.0	3.8	9	0.36
BARBEL	1699	14.9	7.6	37	0.3	207	1455	1.4	13.0	72	4.14
	177	3.3	0.4	15	0.2	39	181	0.3	0.5	7	0.19
BREAM	3109	21.7	7.8	151	2.9	240	2717	2.4	16.3	96	4.09
	700	4.8	0.5	74	1.8	116	701	1.6	4.0	7	0.18

t-Test results:

tBIO

	G	Ba	Br
T	*	ns	ns
G		ns	**
Ba			*

tDENS

	G	Ba	Br
T	**	**	**
G		**	ns
Ba			ns

NofSP

	G	Ba	Br
T	ns	**	**
G		**	*
Ba			ns

salBIO

	G	Ba	Br
T	**	**	**
G		**	ns
Ba			*

salDENS

	G	Ba	Br
T	**	**	**
G		**	ns
Ba			*

eelBIO

	G	Ba	Br
T	ns	**	ns
G		ns	ns
Ba			ns

coa BIO

	G	Ba	Br
T	ns	**	*
G		**	**
Ba			*

eelDENS

	G	Ba	Br
T	ns	**	*
G		*	ns
Ba			ns

coaDENS

	G	Ba	Br
T	ns	ns	ns
G		ns	ns
Ba			ns

BMWP

	G	Ba	Br
T	ns	ns	ns
G		ns	ns
Ba			*

ASPT

	G	Ba	Br
T	ns	**	*
G		*	ns
Ba			ns

* $p<0.05$
** $p<0.01$
ns $p>0.05$

A decline in salmonids and increase in cyprinids from trout zone to bream zone is to be expected from the above descriptions of each zone. The effects of water quality should be seen as being superimposed on this natural distribution. An indication of this is given in Figure 8, where it is evident that the mean total biomass in the grayling zone (1184 g 100 m^{-2}) was significantly depressed compared to that in the trout (2327 g 100 m^{-2}) and bream (3109 g 100 m^{-2}) zones. From Figure 5 it can be seen that the grayling zone had the largest proportion of class 3 sites (33% of those classified, or 25% assuming all 8 Scottish sites in the grayling zone are of class 1A), followed by the barbel zone (25%) which had the second lowest mean total biomass (1699 g 100 m^{-2}).

Figure 9 shows mean total density rapidly declining from the trout zone to the lower zones. As stated before, this is due to the greater effort put into the capture of smaller individuals in salmonid-dominated areas, indicated by the very high total density in the trout zone in relation to total biomass. In spite of apparent water quality effects in the intermediate zones, Figure 10 shows the mean number of fish species (NofSP) increasing from trout (4.5) to bream (7.8) zone, as would be expected from known autecological tolerances (Section 3.2.2). All differences are significant apart from those between trout and grayling zones (ie the two zones of the salmonid region), and barbel and bream zones (the two zones of the cyprinid region).

Regarding invertebrate status, mean ASPT declined significantly from trout (6.34) to bream (4.09) zone, probably largely due to the predominance of finer substrates downstream. BMWP score showed no clear trend with fish zone.

Although the spread of NWC classes between fish zones was not even, particularly in the trout zone where all sites were of class 1A, it is clear that Figures 8 to 10 show strong fundamental influences of physical habitat on fish populations, and if water quality effects are to be positively identified this source of variation needs to be eliminated as far as is possible by investigating relationships between fishery status and water quality within each zone. However, it should be recognised that this approach cannot account for a large proportion of longitudinal variation in fish populations due to the continuous nature of the variation. It also does not attempt to account for site-specific variations in habitat diversity and carrying capacity, which are further habitat effects which will affect the size and composition of fish populations (Section 3.2.2).

<u>The Trout zone</u>

Eleven of the 112 sites in the database lay in the trout zone, all being of NWC class 1A or equivalent (4 sites are in Scotland and thus have no NWC status in this study). Correlations between environmental and biological variables with sufficient data are given in Table 23.

Table 23 - Correlations between fishery status and physico-chemical parameters in the Trout zone

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
MWIDTH	-.35	-.51	.06	-.54	-.48	.07	-.57	.07	-.62[1]
ALT	-.48	-.40	-.76[2]	-.37	.21	-.29	-.63[1]	-.40	-.61[1]
SLOPE	-.17	.10	-.68[1]	.10	.67[1]	-.27	-.24	-.39	-.09
pH	.07	-.18	.48	.04	-.44	-.08	.57	-.01	.30
COND	.26	.14	.50	.35	.06	-.16	.82[2]	-.13	.40
TEMP	.80[2]	.81[2]	.36	.63[1]	.14	.66[1]	.31	.66[1]	.56
SSLDS	.45	.34	.76[2]	.35	-.18	.18	.82[2]	.26	.59
BOD	.36	.09	.42	.07	-.25	.37	.19	.36	-.22
DO	-.56	-.63[1]	-.14	-.60	-.19	-.35	-.27	-.38	-.63[1]
NH_3N	.70[1]	.49	.60	.31	-.28	.72[1]	.31	.77[2]	.10
NO_2N	.79[1]	.65	.71	.15	-.68	.91[2]	.07	.99[3]	.06
NO_3N	.36	.24	.58	.49	.08	-.16	.89[3]	-.11	.47
PO_4	.68[1]	.59	.45	.37	-.30	.66[1]	.37	.79[2]	.44
THARD	.54	.53	.70	.62	-.47	.30	.89[2]	.40	.89[2]
ALKY	.54	.51	.74	.58	-.48	.31	.87[1]	.40	.86[1]

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

The lack of toxic effect from the consistently measured water quality parameters at these sites was evident from the significant positive correlations of NH_3N and NO_2N with total biomass (both at $p<0.05$), eel biomass (at $p<0.05$ and 0.01 respectively) and eel density (at $p<0.01$ and 0.001 respectively). As with the analysis of the Tavy catchment, correlations indicated an increase in population size with increasing nutrient status (and hence productivity). The Tavy sites formed the greater part of this subset (7 sites out of 11). There was still a negative altitude ($p<0.01$) and slope effect ($p<0.05$) on the number of fish species present, and a negative altitude effect on coarse fish biomass and density, indicating that a significant residual longitudinal habitat effect remained after zonation.

Mean temperature was negatively correlated with total biomass ($p<0.01$), total density ($p<0.01$), salmonid biomass ($p<0.05$), eel density ($p<0.05$) and biomass ($p<0.05$). These correlations may indicate a real effect, since low temperature may be limiting production in the most upland of sites.

Since no likely toxic effect was indicated in this zone by univariate correlations, no toxicity or MRA was undertaken. The variation in fishery status with NWC class could not be investigated within the trout zone, since all sites belonged to class 1A or equivalent.

<u>The Grayling zone</u>

Thirty-two sites lay within the Grayling zone, 8 of which were in Scotland and therefore have no NWC classification in this study (although 6 are the equivalent of class 1A sites). The spread of the remaining 24 sites by NWC class is given below and in Figure 5.

CLASS	1A	1B	2	3
NO. SITES	3	7	6	8

Correlations between environmental and biological variables with sufficient data are given in Table 24.

Significant negative correlations were evident between the NofSP and both NH_3N ($p<0.01$) and NO_2N ($p<0.05$), salmonid biomass and NO_2N ($p<0.05$), and salmonid density and NO_2N ($p<0.05$). NH_3N values ranged from 0.02 to 3.87 mg l^{-1} and at such concentrations are very likely to have a real impact on fish. NO_2N concentrations were relatively low, ranging from 0.02 to 0.38 mg l^{-1}, and correlations with fishery parameters are therefore likely to be due to intercorrelation with other influencing factors such as NH_3N.

Table 24 - Correlations between fishery status, invertebrate status and physico-chemical parameters in the Grayling zone

(Numbers are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP	ASPT
MWIDTH	-.07	.13	.03	.32	.25	.01	-.23	.12	-.20	.14	.60
MDEPTH	-.06	-.71[2]	.26	.26	-.69[2]	-.64[1]	.11	-.52	-.44	-.37	
ALT	.01	-.01	-.21	.41[1]	.59[3]	.08	-.20	-.06	-.50[1]	.01	.46
SLOPE	.19	.00	-.06	.14	.11	.06	.14	-.08	.28	.06	.17
pH	.36	.23	.12	-.12	.03	.30	.35	.39	-.06	-.75[2]	-.42
COND	.40	-.28	.32	-.09	-.45	-.23	.57[2]	-.18	.00	-.58	-.74
TEMP	-.07	.02	-.07	-.29	-.14	-.03	.06	.10	.02	-.20	-.66[1]
SSLDS	.00	.46	-.08	-.09	.21	.48[1]	-.17	.57[1]	.06	-.04	.45
BOD	-.13	-.22	-.11	-.38[1]	-.39[1]	-.19	.06	-.19	.13	-.55	-.89[3]
DO	.44[1]	.17	.43[1]	.56[2]	.28	.13	.26	.03	.14	.25	.65[1]
NH_3N	-.36	-.33	-.54[2]	-.32	-.34	-.20	-.23	-.14	-.25	-.53	-.73[1]
NO_2N	-.35	-.59	-.55[1]	-.50[1]	-.59[1]	-.16	-.26	-.18	-.52	-.52	-.75
NO_3N	-.21	-.12	.18	-.59[2]	-.57[2]	-.29	.12	-.09	.50	-.36	-.80[2]
PO_4	-.23	-.37	-.14	-.12	-.42	-.27	.10	-.23	-.09	-.48	-.73
THARD	-.44	-.42	-.23	-.74[1]	-.84[2]	.34	-.08	.46	-.24		
ALKY	.12	-.63	-.21	-.73[2]	-.75[2]	.30	.27	.45	-.46	-.70	-.85[1]
BMWP	.22	.31	.56[1]	.49	.29	.00	.00	-.07	.64[1]		.55
ASPT	.17	.27	.51	.60	.71[1]	.09	-.23	-.03	-.03		

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

Positive correlations were evident between DO and total fish biomass ($p<0.05$), NofSP ($p<0.05$), and salmonid biomass ($p<0.01$). The lowest mean DO in the grayling zone was only 8.2 mg l^{-1}, and would not be expected to affect fish unless large fluctuations were occurring. The mandatory standard for EC designated salmonid waters is 50% exceedance of 9 mg l^{-1} (see Appendix A.1). The site with the lowest mean DO, site S07, had a median value of 9.2 mg l^{-1}, although the standard deviation about the mean was 2.55 and the lowest value recorded was 4.4 mg l^{-1}. A real effect of DO on fisheries in the grayling zone can therefore not be ruled out.

Salmonid density was positively correlated with altitude ($p<0.001$) and negatively correlated with mean depth ($p<0.01$), and salmonid biomass was positively correlated with altitude ($p<0.05$). As previously stated, the correlations with salmonid density are likely to be mainly due to sampling artefacts (see earlier in this section). However, this may well be compounded with residual longitudinal variation in habitat and also water quality effects, particularly from NH_3N. The correlation of salmonid biomass with altitude is likely to be due to either or both of the latter two effects. Salmonid biomass was also negatively correlated ($p<0.05$) with BOD, which may be indicating a sediment quality problem associated with egg development.

Regarding invertebrate status, ASPT was positively correlated with salmonid density ($p<0.05$), suggesting some sort of real environmental influence on the latter. ASPT was also negatively correlated with NH_3N ($p<0.05$) and BOD ($p<0.001$), and positively correlated with DO. BOD does not have a toxic effect in itself but may be acting as a surrogate for sediment quality, since the settlement of particulates with a high BOD is likely to result in low interstitial DO, even if the DO in the overlying water is adequate. These correlations suggest that water quality, and possibly sediment quality, is having a discernable effect on the distribution of sensitive invertebrates in this zone.

Table 25 shows correlations between fishery parameters and toxicity scores, as calculated by the TOXIC program.

Table 25 - Correlations between fishery status, invertebrate status and toxicity scores in the Grayling zone

(Values are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
TOXT1A	-.48[1]	-.19	-.62[2]	-.22	.16	-.19	-.45[1]	-.17	-.29
TOXT3A	-.44[1]	-.48	-.63[2]	-.44[1]	-.46[1]	-.10	-.33	.09	-.40
TOXR1A	-.37	-.39	-.60[2]			-.13	-.29	-.16	-.35
TOXR3A	-.33	-.35	-.57[2]			-.10	-.25	-.03	-.31
TOXR1T	-.42[1]	-.42	-.64[2]			-.15	-.33	-.22	-.39
TOXR3T	-.36	-.39	-.60[2]			-.11	-.28	-.08	-.34

NoFSP was negativly correlated ($p<0.01$) with all 6 toxicity scores. Since no habitat effect on NoFSP was indicated previously (see Table 24), these correlations would appear to be solely attributable to toxicity. This is reinforced by the significant correlations between NoFSP and NH_3N, NO_2N and DO evident in Table 24.

Coarse fish biomass was negatively correlated with TOXT1A ($p<0.05$), which may be indicating a significant toxic effect due to metals since no correlation was evident between coarse fish biomass and TOXT3A.

Salmonid biomass and density were negatively correlated ($p<0.05$) with TOXT3A, whilst total biomass was negatively correlated (all at $p<0.05$) with TOXT3A, TOXT1A and TOXR1T. There were no correlations between either eel biomass or density and toxicity scores.

It is clear that the trout toxicity scores are better correlated with fishery parameters than roach scores, probably due to the better description of toxic effect by the latter.

Table 26 shows the results of MRA, using those environmental variables for which a possible causative influence was detected in the correlation analysis. Stepwise multiple regression was used where predictor variables did not exhibit a significant intercorrelation; alternatively, a single R-squared value has been given for a set of variables.

Table 26 - Multiple regression analysis of sites in the Grayling zone

(Values are R-squared values, indicating the percentage of total variance accounted for).

	NofSP		tot BIOMASS
NH_3N, DO, NO_2N:	49.7%	DO, NH_3N, NO_2N:	22.5%
TOXT3A:	39.9%	TOXT3A:	19.1%
TOXT1A:	38.0%	TOXT1A:	22.9%
(ALT, SLOPE:	3.6%)		
	salBIOMASS		**coa BIOMASS**
DO, NO_2N, ALT, BOD, NH_3N:	30.7%	TOXT1A:	20.0%
ALT, BOD, TOXT3A:	29.3%		

NH_3N, NO_2N and DO combined, explained 49.7% of the variance in the number of species. The low explanation of variance by altitude and slope (3.6%) indicates that this result is due to a toxic effect, rather than an intercorrelation with habitat parameters. Due to inter-correlations between NH_3N, DO and NO_2N the relative effects of these parameters cannot be separated by MRA. When these 3 variables were combined into TOXT3A the amount of explained variance in NoFSP was reduced to 39.9%, indicating that raw water quality data is describing the variance in NoFSP better than toxicity-weighted data. When the available metals toxicity data was added in by replacing TOXT3A by TOXT1A the explained variance was reduced again to 38.0%.

DO, NH_3N and NO_2N combined, explained 22.5% of the variance in total biomass. This was reduced to 19.1% by replacing these predictors by TOXT3A, and marginally increased by replacing them with TOXT1A.

NO_2N, altitude, DO, NH_3N and BOD explained 30.7% of the variance in salmonid biomass. This was marginally reduced to 29.3% by substituting

TOXT3A for DO, BOD and NH_3N. TOXT1A accounted for 20.0% of the variance in coarse fish biomass.

Table 27 and Figures 11, 12 and 13 show the variation in fishery status with NWC class within the grayling zone.

Table 27 - Fishery status by NWC class in the Grayling zone

(Mean values with standard error below each mean. Biomass values in g 100 m^{-2}, density values in No. 100 m^{-2}).

NWC	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
1A	1546	21.7	5.0	1532	15.4	0	14	0.0	6.3
	363	8.4	0.0	376	2.1	0	14	0.0	6.3
1B	1827	27.5	7.0	585	6.2	66	1176	1.0	20.3
	815	11.4	1.3	162	2.4	43	781	0.7	9.0
2	905	21.9	8.7	9	0.1	41	854	0.4	21.3
	319	16.4	1.4	9	0.1	17	309	0.2	16.0
3	673	0.9	2.4	14	0.0	220	439	0.0	0.9
	670	0.7	1.1	14	0.0	220	436	0.0	0.7

t-Test results:

tBIO

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

tDENS

	1B	2	3
1A	ns	ns	*
1B		ns	ns
2			ns

NOfSP

	1B	2	3
1A	ns	ns	ns
1B		ns	*
2			**

salBIO

	1B	2	3
1A	*	**	**
1B		*	**
2			ns

salDENS

	1B	2	3
1A	ns	**	**
1B		*	*
2			ns

eelBIO

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

coaBIO

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

eelDENS

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

coaDENS

	1B	2	3
1A	ns	ns	ns
1B		ns	ns
2			ns

* $p<0.05$
** $p<0.01$
ns not significant

Figure 11. Mean fish biomass by NWC class in the Grayling zone.

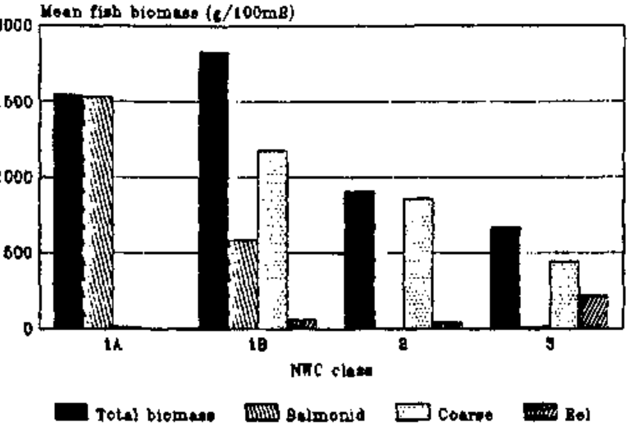


Figure 12. Mean fish density by NWC class in the Grayling zone.

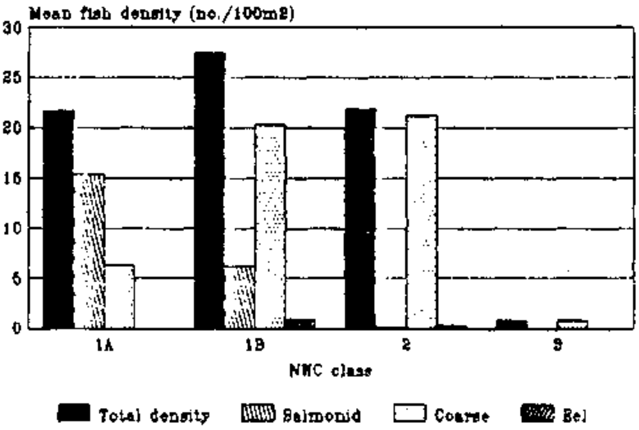
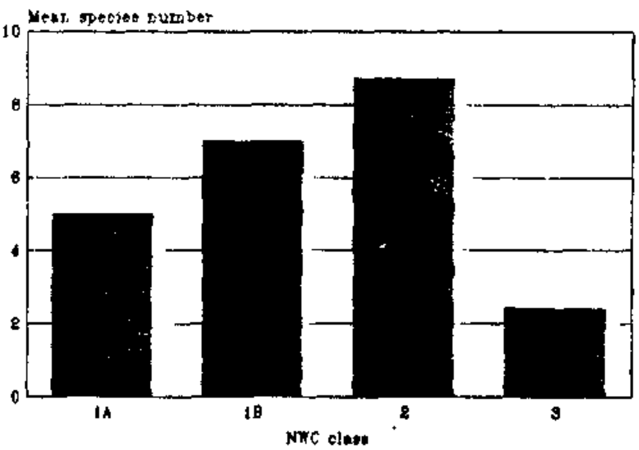


Figure 13. Mean number of fish species by NWC class in the Grayling zone.



Mean salmonid biomass showed a significant decline from class 1A down to classes 2 and 3 which, as seen above, may be partly due to NH_3N but is also likely to be influenced by residual habitat effects. The importance of the latter is demonstrated by the very low biomass of coarse fish, and also low number of fish species, in class 1A (14 g 100 m^{-2} and 5.0 respectively) compared to class 1B (1176 g 100 m^{-2} and 7.0 respectively), indicating that the 1A sites were typical salmonid habitats, ie at the upstream end of the Grayling zone.

From class 1B to class 3, total biomass and coarse fish biomass showed steady declines, although none of the differences were significant due to the high degree of variability in the data. Coarse fish density (Figure 3) showed a drastic decline from class 2 (21.3 fish 100 m^{-2}) to class 3 (0.9 fish 100 m^{-2}), although again this is not statistically significant. This decline may reflect an increased proportion of older fish at class 3 sites, which would be the case if spawning success were being inhibited by impoverished water quality. Salmonid density reflected the decline in salmonid biomass with NWC class.

The Barbel zone

Forty-eight sites lay within the Barbel zone, and are distributed by NWC class as follows (see also Figure 5):

CLASS	1A	1B	2	3
No. OF SITES	0	19	17	12

Correlations between environmental and biological variables with sufficient data are given in Table 28.

As might be expected from natural longitudinal fish distributions, salmonids were virtually absent from the Barbel zone, with a mean biomass of 37 g 100 m^{-2} and a mean density of 0.3 fish 100 m^{-2} (see Table 22). Coarse fish biomass and total biomass were not significantly correlated with any consistently measured environmental variable.

Highly significant negative correlations were evident between total fish density and both alkalinity ($p<0.001$) and hardness ($p<0.001$), between number of fish species and alkalinity ($p<0.001$), and between coarse fish density and both total hardness ($p<0.001$) and alkalinity ($p<0.001$). Under normal conditions fishery status might be expected to increase with increases in hardness and alkalinity, so it is possible that one or more water quality parameters are having a toxic effect. NH_3N would not appear to have had a major influence since it was not significantly negatively correlated with any fishery parameter other than mean eel density. However, the highest NH_3N concentration within the zone was 2.06 mg l^{-1} (SD = 1.65), at site TH8, which should be sufficient to have an effect on fish, depending upon the distribution between the un-ionised and ionised forms.

Table 28 - Correlations between fishery status, invertebrate status and physico-chemical parameters in the Barbel zone

(Values are Pearson coefficients of correlation)

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP	ASPT
MWIDTH	-.27	-.51[2]	.04	-.32[1]	-.27	-.31[1]	-.18	-.47[1]	-.48[1]	.18	.02
MDEPTH	.29	-.06	-.03	.14	.05	.13	.22	.11	-.10	.60[2]	-
ALT	-.05	-.30	.05	-.26	-.14	-.12	.00	-.27	-.29	-.33	-.76[2]
SLOPE	.15	.57[3]	.07	.41[2]	.35[1]	.07	.09	.29	.56[3]	-.22	-.44
pH	-.01	-.39[1]	-.02	-.18	-.34[1]	.05	.00	.04	-.39[1]	.34	.76[2]
COND	-.13	-.32	.08	-.40[2]	-.38[1]	.16	-.13	.30	-.33	-.16	-
TEMP	-.04	-.05	-.13	-.10	-.13	-.17	.01	.03	-.04	.00	-.43
SSLDS	-.22	-.18	-.45[2]	-.14	-.12	.45[2]	-.30	.32	-.21	-.29	-.63
BOD	.13	-.01	.08	-.21	-.22	.21	.10	.43[2]	-.03	-.48[1]	-.30
DO	.24	-.01	.16	.14	-.04	.08	.21	.01	-.01	.01	.32
NH_3N	-.23	-.14	-.12	-.21	-.16	.24	-.26	.35[1]	-.17	-.44[1]	-.20
NO_2N	-.27	-.45	-.21	-.34	-.35	-.29	-.20	-.11	-.45	-.41	-.05
NO_3N	.04	-.49	.11	-.29	-.15	-.38	.12	-.65	-.48	-.13	.27
PO_4	-.21	-.32	-.11	-.24	-.28	.01	-.20	.44	-.34	-.61	-.45
THARD	-.34	-.83[3]	-.34	-.60[2]	-.63[2]	.09	-.28	.02	-.84[3]	.27	
ALKY	-.31	-.83[3]	-.62[3]	-.34	-.57[2]	.03	-.27	-.11	-.83[3]	.41	
BMWP	.03	-.28	.04	.15	-.03	-.17	.05	-.38	-.23	1.00	.90[3]
ASPT	-.01	.64	-.01	.24		.70[1]	-.16	.80	.61		

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

BMWP score and ASPT were not significantly correlated with any fishery parameter other than eel biomass.

Correlations between fishery parameters and toxicity scores are given in Table 29.

Table 29 - Correlations between fishery status and toxicity scores in the Barbel zone

(Values are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
TOXT1A	-.24	-.07	-.20	-.09	-.07	.03	-.23	.24	-.08
TOXT3A	-.15	.00	-.33[1]	-.03	-.03	-.02	-.14	.21	-.02
TOXR1A	-.24	-.19	-.11			.17	-.25	.33	-.22
TOXR3A	-.17	-.16	-.22			.22	-.19	.41[1]	-.19
TOXR1T	-.28	-.18	-.09			.09	-.27	.26	-.20
TOXR3T	-.19	-.13	-.26			.16	-.20	.38[1]	-.16

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

Only one negative correlation was significant, between the number of species and TOXT3A ($p<0.05$). Eel density was positively correlated ($p<0.05$) with both TOXR3A and TOXR3T; it was also positively correlated with NH_3N (see Table 28).

The range of mean toxicity scores within the zone is given below:

TOXT1A	0 - 9	TOXR1A	0 - 17	TOXR1T	0 - 20
TOXT3A	0 - 9	TOXR3A	0 - 17	TOXR3T	0 - 20

Since each TOX1 score has the same range as the equivalent TOX3 score it can be inferred that all measured toxicity is due to either NH_3N, DO or NO_2N. Toxic effects might be expected at sites with mean toxicity scores as high as the maximum roach toxicity values. However, these values may be spuriously high for reasons mentioned previously, such that trout scores may be giving more realistic results.

MRA was performed on variables where correlations showed a likely toxic effect, and the results are given in Table 30.

Table 30 - Multiple regression analysis of sites in the Barbel zone

(Values are R-squared values, indicating the percentage of total variance accounted for).

	NofSP		tBIOMASS
TOXT3A:	11.1%	NH_3N, DO:	7.7%
DO, NH_3N, NO_2N:	22.4%		

TOXT3A accounted for 11.1% of the variance in the number of species, but when it was replaced with its constituent raw variables (DO, NH_3N and NO_2N) this value doubled to 22.4%; however, only 12 observations were available in the latter regression. NH_3N and mean DO only accounted for 7.7% of the variance in total biomass.

Table 31 and Figures 14, 15 and 16 show the variation in fishery and invertebrate status with NWC class.

The significant increase in mean total biomass ($p<0.05$) from class 1B (1257g 100 m^{-2}) to class 2 (2157g 100 m^{-2}) would appear to indicate that NWC class status is not reflecting a major influence on fish populations. However, this may stem from a number of confounding influences, discussed in Section 3.2.2, and is not necessarily related to water quality. The difference in total biomass between the two classes was almost entirely due to variation in coarse fish biomass. It is possible that some slow-water cyprinids may be constrained by current velocity, which may be

Table 31 - Fishery and invertebrate status by NWC class in the Barbel zone

(Mean values with standard error below each mean. Biomass values in g 100 m^{-2}, density values in No. 100 m^{-2}).

NWC	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP	ASPT
1B	1257	17.9	6.7	79.3	0.8	240	937	1.6	15.4	83	4.2
	245	7.2	0.6	7.6	0.6	59	233	0.3	1.7	11	0.3
2	2157	8.9	9.2	0.0	0.0	90	2066	0.3	8.6	73	4.4
	332	1.5	0.6	0.0	0.0	50	325	0.2	1.4	8	0.2
3	1752	18.0	7.2	23.8	0.0	320	1409	2.8	15.2	61	3.7
	311	4.4	0.7	22.8	0.0	94	339	0.6	4.2	14	0.5

t-Test results:

tBIO

	2	3
1B	*	ns
2		ns

tDENS

	2	3
1B	ns	ns
2		*

NOfSP

	2	3
1B	*	ns
2		*

salBIO

	2	3
1B	*	ns
2		ns

salDENS

	2	3
1B	ns	ns
2		ns

eelBIO

	2	3
1B	ns	ns
2		*

coaBIO

	2	3
1B	*	ns
2		ns

eelDENS

	2	3
1B	**	ns
2		**

coaDENS

	2	3
1B	ns	ns
2		ns

BMWP

	2	3
1B	ns	ns
2		ns

ASPT

	2	3
1B	ns	ns
2		ns

* $p<0.05$
** $p<0.01$
ns not significant

generally stronger at class 1B sites. This possibility is reinforced by the significant increase ($p<0.05$) in the NofSP from class 1B (6.7) to 2 (9.2). However, current velocity should not affect total biomass, since each site has a certain production potential irrespective of the number and nature of species exploiting that potential.

There were no significant differences evident in BMWP score or ASPT between NWC classes to shed light on the reduced fish biomass evident in class 1B.

Figure 14. Mean fish biomass by NWC class in the Barbel zone.

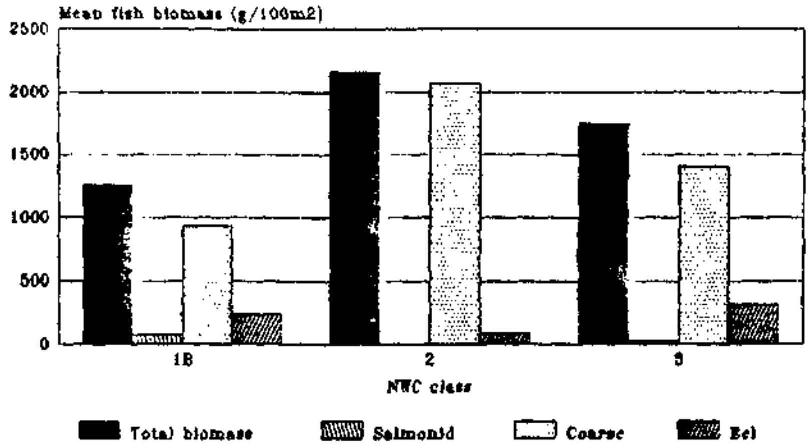


Figure 15. Mean fish density by NWC class in the Barbel zone.

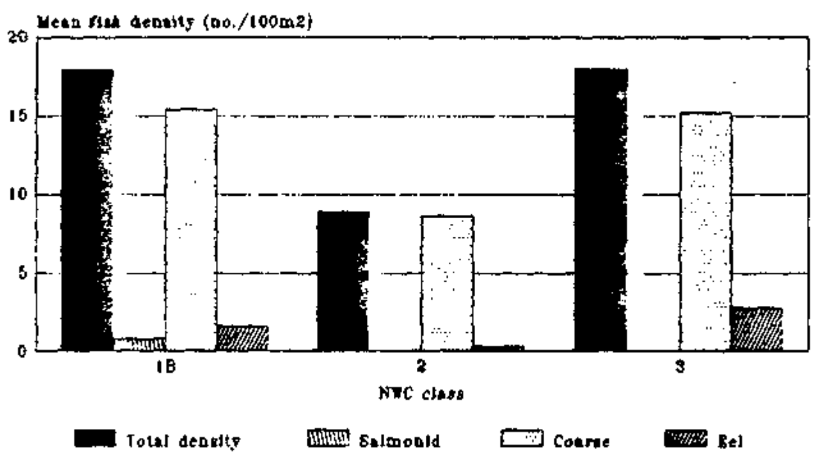
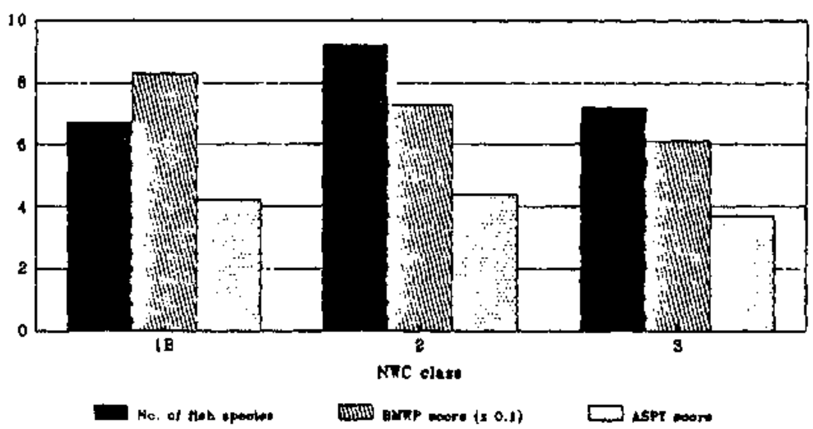


Figure 16. Mean number of fish species and invertebrate scores by NWC class in the Barbel zone.



The Bream zone

Nineteen sites lay in the Bream zone, being distributed between NWC classes as follows (see also Figure 5):

CLASS	1A	1B	2	3
NO OF SITES	0	10	6	3

Correlations for those environmental and biological variables with sufficient data are given in Table 32.

Table 32 - Correlations between fishery status, invertebrate status and physico-chemical parameters in the Bream zone

(Values are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS	BMWP
MWIDTH	-.44	-.49[1]	-.20	-.15	-.15	-.24	-.38	-.41	-.36	.15
MDEPTH	-.07	.31	.73[1]	-.14	.16	-.46	-.02	-.39	.27	-.46
ALT	.18	.10	.05	.42	.33	-.31	.19	-.35	.12	-.02
SLOPE	.43	-.03	.24	-.09	-.07	.27	.39	.15	-.05	.02
pH	-.04	-.40	-.30	.12	.09	-.46[1]	.03	-.52[1]	-.33	.30
COND	-.05	-.60	.12	-.41	-.31	-.40	.07	-.35	-.43	.07
TEMP	.16	.26	-.04	-.45	-.34	-.03	.21	.08	.44	-.45
SSLDS	-.53	-.11	-.01	.44	.31	-.05	-.58[1]	.74[2]	-.35	.24
BOD	.03	.22	.10	-.21	-.16	-.15	.08	-.17	.40	-.25
DO	-.01	-.34	.18	.25	.18	.13	-.06	.04	-.52[1]	.67[1]
NH_3N	.03	.37	.10	.05	.04	-.20	.06	-.17	.49[1]	-.67[1]
NO_2N	-.29		-.51	-.81	-.81	-.82[1]	-.13			
NO_3N	-.63		-.32	-.02	-.02	-.10	-.62			
PO_4	.34	.05	.39	-.38	-.28	-.33	.42	-.23	.39	.07
THARD	-.08	-.67[1]	-.21	.10	.09	-.78[1]	.05	-.80[2]	-.46	.48
ALKY	-.33	-.08	-.34	.81[3]	.69[2]	-.52	-.32	-.51	-.25	-.13
BMWP	-.31	-.83[2]	-.55	-.03	-.21	-.22	-.24	-.25	-.77[2]	

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

As in the case of the Barbel zone, salmonids were present at very low densities and biomass in this zone, with means of 2.9 100 m^{-2} and 157g 100 m^{-2} respectively (see Table 22). Coarse fish density was positively correlated ($p<0.05$) with NH_3N and negatively correlated ($p<0.05$) with DO, and it would therefore appear that these variables did not having a detrimental effect on fisheries at these sites. However, the lowest mean DO in this zone was 6.14 mg l^{-1} (standard deviation = 4.0), below the mandatory 50% exceedance level of 7 mg l^{-1} for EC designated cyprinid waters (see Appendix A.1). This value might reasonably be expected to have an impact on fish distribution, especially in view of the large variability around the mean. However, on the day of the fish survey at the site in question (TH5a), DO was very low at 3.8 mg l^{-1} yet the fish yield was very high, at 4519 g 100 m^{-2}. TH5a also has the highest NH_3N concentration in the Barbel zone, at 1.83 mg l^{-1} (SD = 1.45). This is well above the mandatory 95% compliance level of 0.78 mg l^{-1} for EC designated waters (see Appendix A.1). On the day of the fish survey at TH5a the total NH_3N concentration was much higher, at 3.8 mg l^{-1}.

NofSP was positively correlated ($p<0.05$) with mean depth, as might be expected at sites supporting slow/still-water cyprinids. Coarse fish biomass was negatively correlated ($p<0.05$) with SSLDS which ranged from 6.1 to 49.4 mg l^{-1}; Alabaster and Lloyd (1982) concluded that average suspended solids levels of >25 mg l^{-1} could reduce fish yield, and the UK standard for EC designated waters is consequently set at this level (see Appendix A.1). Eel biomass was negatively correlated ($p<0.05$) with NO_2N, which ranged between 0.03 and 0.12 mg l^{-1}, concentrations too low to be producing a toxic effect.

BMWP score was negatively correlated ($p<0.05$) with NH_3N and positively correlated ($p<0.05$) with DO, in contrast to the correlations between the same variables and coarse fish density. As might be expected from this, BMWP score was negatively correlated ($p<0.01$) with coarse fish density, and consequently total density. Considering the lowest DO and highest NH_3N in this zone, the correlations with BMWP score may well be due to a toxic effect that the resident fish populations have not responded to.

Table 33 gives correlations between fishery parameters and toxicity scores.

Table 33 - Correlations between fishery status and toxicity scores in the Bream zone

(Values are Pearson coefficients of correlation).

	tBIO	tDENS	NofSP	salBIO	salDENS	eelBIO	coaBIO	eelDENS	coaDENS
TOXT1A	-.17	-.25	.13	-.15	-.12	-.36	-.09	-.30	-.12
TOXT3A	-.30	.20	.06	.28	.22	-.26	-.29	-.24	.18
TOXR1A	.00	-.21	.10			-.30	.05	-.26	-.11
TOXR3A	-.15	-.23	.19			-.59[1]	-.09	-.58[1]	-.20
TOXR1T	-.16	-.35	-.10			-.27	-.09	-.24	-.24
TOXR3T	-.36	-.35	.05			-.65[2]	-.28	-.68[2]	-.26

The only significant correlations were between eel biomass and both mean TOXR3A ($p<0.05$) and TOXR3T ($p<0.01$), and eel density and both TOXR3A ($p<0.05$) and TOXR3T (0.01). The range of means of each toxicity score within the zone is given below:

TOXT1A	0 - 4	TOXR1A	0 - 13	TOXR1T	0 - 21
TOXT3A	0 - 2	TOXR3A	0 - 3	TOXR3T	0 - 3

Trout toxicity scores indicate no real toxicity, and because these scores describe variations in toxicity more accurately than the roach scores these are probably a better measure of toxicity for cyprinids as well as salmonids. The differences between maximum TOXR1 and TOXR3 scores appears to indicate that metals should have a toxic effect within the zone; however, univariate correlations between these scores and fishery parameters do not support this. No account of variations in metal toxicity due to hardness is taken in the calculation of roach toxicity scores, and since the Bream zone generally consists of hard waters the toxicity may be over-estimated.

Table 34 gives the results of MRA, using information from univariate correlation analysis as a guide.

Table 34 - Multiple regression analysis of sites in the Bream zone

(Values are R-squared values, indicating the percentage of total variance accounted for).

EEL BIOMASS		EEL DENSITY	
TOXR3A:	39.0%	TOXR3A:	33.7%
TOXR3T:	42.4%	TOXR3T:	46.6%
NO_2N:	67.7%	NO_2N:	65.8%
DO, NH_3N:	3.8%	DO, NH_3N:	4.4%
SSLDS:	0.3%	SSLDS:	54.9%
COARSE BIOMASS			
SSLDS:	33.8%		
SSLDS, NH_3N, DO:	45.8%		

Roach toxicity scores explained a large amount of the variance in eel biomass and density. When these scores were broken down into their constituent variables it was apparent that NO_2N explained 67.7% and 65.8% of the variance in eel biomass and density respectively. However, NO_2N concentrations are low in the catchment, as mentioned previously, and it is probable that it is acting as a marker for another variable. SSLDS accounted for 54.9% of the variance in eel density, but did not account for any variance in eel biomass.

A large proportion (33.8%) of the variance in coarse fish biomass was explained by mean SSLDS, and this was increased to 45.8% when SSLDS, NH_3N and DO were used in combination. Figure 17 shows the relationship between coarse fish biomass and mean SSLDS in the Bream zone. No biomass greater than 2500 g 100 m^{-2} is evident over 16 mg l^{-1} SSLDS.

The increase in variance of fish biomass with decreasing SSLDS might be expected, since as one constraint on population growth is relieved another will take over. This may be another water quality or a habitat constraint.

Too few sites were available in the Bream zone to investigate variations in fishery and invertebrate status with NWC class.

Analyses of factors affecting selected species

Analyses were performed on the measured biomass of selected fish species, with the database divided into the Salmonid (Trout and Grayling zones) and Cyprinid (Barbel and Bream zones) regions. Of all the sites 38% were designated as in the salmonid region, 60% in the Cyprinid region and 2% were not designated. Only species with a significant number (>15) of positive biomass records (ie >0 g 100 m^{-2}) in the region relevant to the species were used. The species thus selected were trout, salmon, pike, perch, chub, dace, roach and gudgeon. The biomass records for these species by site are given in Appendix F. The distribution of biomass records by region is summarised in Table 35.

Table 35 - The distribution of biomass records for selected species

Fish Species	No of Positive Biomass Records	% Positive Biomass Records in Region	
		Salmonid Region	Cyprinid Region
trout	42	67	33
salmon	21	81	19
pike	48	10	90
perch	46	20	80
chub	59	15	85
dace	58	10	90
roach	61	16	84
gudgeon	64	16	84
eels	70	39	61

Eels were the most often recorded and also the most ubiquitous species, often being recorded in the salmonid region. Eels have been discussed elsewhere and are not analysed further here. Gudgeon, roach, chub and dace were the next most recorded fish and were found predominantly in the Cyprinid region. The remaining species were generally true to their ascribed regions, except trout which were found in the Cyprinid region for 33% of the total biomass records (though there were more sites in the Cyprinid region than the salmonid region). Trout and salmon were absent from 33% and 60%, respectively, of the designated Salmonid region sites. Pike, perch, chub, dace, roach and gudgeon were absent from 24%, 45%, 26%, 24%, 22% and 19%, respectively, of the Cyprinid region.

Table 36 gives correlations between species biomass, selected physico-chemical parameters and toxic scores. From this matrix MRA was performed to determine the importance of variables in contributing to the total variance.

Trout

Trout biomass was significantly positively correlated with slope and negatively correlated with SSLDS, NH_3N, NO_2N and sTOXT3A (Table 36). MRA identified slope and nitrite as major explanatory variables, together explaining 53% (R-squared) of the variance. The inclusion of suspended solids and ammonia increased the variance to 57%, but both of these parameters were intercorrelated with NO_2N. The inclusion of NO_2N reduced the number of observations in the analysis due to its relative scarcity in the database. Regression using slope, SSLDS and NH_3N (the latter two water quality parameters not being significantly intercorrelated) only explained 39% of the variance. Regression using slope and sTOXTR3A explained 44% of the variance. The addition of SSLDS (a variable not included in the TOXIC program) did not explain more of the variance.

The implications of these results is that habitat is indicated as a major factor influencing trout biomass, even within the salmonid region. This has been borne out by the relatively successful use of habitat

Table 36 – Correlations between the biomass of selected fish species with water chemistry, habitat parameters and toxicity scores

(Values are Pearson coefficients of correlation).

Salmonid region:

	sTroutBIO	sSalmonBIO
sWIDTH	-0.21	0.14
sDEPTH	-0.01	-0.63[2]
sALT	0.11	0.31[1]
sSLOPE	0.55[3]	0.43[2]
sSSLDS	-0.36[1]	-0.07
sDO	0.26	0.15
sNH_3N	-0.33[1]	-0.27
sNO_2N	-0.59[2]	-0.48[1]
sTOXT1A	-0.01	0.28
sTOXT3A	-0.46[1]	-0.31

Cyprinid region:

	cPikeBIO	cPerchBIO	cChubBIO	cDaceBIO	cRoachBIO	cGudgBIO
cWIDTH	-0.01	-0.13	-0.23	-0.24	-0.04	-0.31[1]
cDEPTH	0.23	-0.01	0.01	-0.09	0.30[1]	-0.32
cALT	-0.09	-0.06	0.05	-0.08	0.20	-0.06
cSLOPE	-0.26[1]	0.14	-0.01	0.05	-0.13	0.26[1]
cSSLDS	-0.20	-0.13	-0.26[1]	-0.27[2]	-0.25[1]	-0.09
cDO	-0.16	-0.08	0.20	0.33[2]	-0.32[2]	0.31[1]
cNH_3N	-0.15	-0.11	-0.17	-0.18	-0.01	0.33[1]
cNO_2N	-0.22	-0.18	-0.15	-0.19	-0.15	0.46
cTOXT1A	-0.12	0.06	-0.14	-0.19	-0.12	0.32[1]
cTOXT3A	-0.13	-0.00	-0.18	-0.24	-0.09	0.20
cTOXR1A	-0.13	0.02	-0.10	-0.03	-0.14	0.40[2]
cTOXR3A	-0.15	0.06	-0.16	-0.08	-0.15	0.39[2]
cTOXR1T	-0.13	-0.00	-0.09	-0.07	-0.14	0.28[1]
cTOXR3T	-0.16	-0.09	-0.17	-0.11	-0.15	0.37[2]

[1] $p<0.05$
[2] $p<0.01$
[3] $p<0.001$

s = salmonid region subset
c = cyprinid region subset

evaluation procedures to predict trout abundance in salmonid streams (Milner et al 1985). Nitrite may be an indicator of sewage pollution an acting as a surrogate for SSLDS, NH_3N or some other effect not identified in the analysis.

Salmon

The correlation matrix (Table 36) identifies water depth and NO_2N as parameters negatively correlated with salmon biomass, and altitude and slope as those positively correlated. MRA combining the two more significant variables (depth and NO_2N) was not possible, due to a shortage of data. In isolation the variables accounted for 39% and 23% of the variance respectively. Depth and altitude together explained 46% of the variance. Slope was intercorrelated with these and did not contribute further to the variance. There was insufficient data to use any toxicity scores.

Again habitat (though different parameters) and NO_2N were important variables. Depth has been identified as an important factor in studies of juvenile salmon.

Pike

The correlation matrix (Table 36) demonstrated only a poorly significant ($p<0.05$) negative correlation with slope. In MRA little variance in pike biomass could be explained by water quality, habitat or toxicity scores. The combination of depth, slope, SSLDS and cTOXR1A explained 21% of the variance, and that of slope and NO_2N explained 15%.

The results appear to indicate an independence of the species from the parameters included in the database at the levels measured. The occurence of backwaters, weed cover and suitable prey species may be more pertinent to the success of the fish. Biomass may not be the most suitable measure of abundance of the fish as single fish may contribute much to the measure and may also be missed in sampling small areas.

Perch

The correlation matrix (Table 36) shows no significant correlations. Regression analyses reflected this and no more than 10% of the variance could not be sensibly be explained without a considerably reduced dataset.

The results indicate that the biomass of this species is, like that of pike, independent of the variables as measured in this database at the levels recorded.

Chub

The correlation matrix (Table 36) shows a negative but poor correlation between fish biomass and SSLDS. The combination of width, SSLDS and DO explained only 15% of the variance. The addition of NH_3N explained a further 1.5%.

The biomass of this species is again not dependent upon the variables measured in this database at the magnitudes recorded and under the sampling regime used. Smith (pers comm) found in studies in East Anglian rivers that the densities of chub were related to river velocity and bankside tree cover, variables not included in this study (although slope was used as a surrogate for the former).

Dace

Dace biomass was significantly ($p<0.01$) negatively correlated with both SSLDS and DO (Table 36), possibly indicating a detrimental impact due to organic pollution. These two parameters accounted for 25% of the variance. The addition of width accounted for a further 3%.

Roach

Roach biomass was positively correlated with depth and negatively correlated with SSLDS and DO (Table 36). MRA using depth and dissolved oxygen explained as much (25%) of the variance as when SSLDS was included. The addition of altitude explained a further 8%. The relationship was a positive one though there was not a significant correlation at $p<0.05$. The substitution of DO by cTOX3RT reduced the explained variance to 25%.

The species seemed to be influenced by both habitat and water quality variables. Depth may be a surrogate for the effect of flow, and more particularly water velocity, the species being more associated with slower flowing rivers than dace and chub, for instance. The contribution by altitude is curious and even contradictory. It may indicate more suitable conditions for the fish in more upstream reaches within the Cyprinid region, due to either preferred water quality or habitat. Univariate correlations suggest SSLDS and both MRA and correlation analysis suggest DO are relevant factors.

Gudgeon

The biomass of gudgeon was positively correlated with slope, DO, ammonia and the toxicity scores cTOX1TA, cTOX1RA, cTOX3RA, cTOX1RT and cTOX3RT (Table 36). Biomass was negatively correlated with width. MRA on a reduced database indicated that NO_2N contributed 20% of the variance alone, but owing to the lack of nitrite records (only 18 of the 57 sites for which gudgeon biomass was recorded) the correlation was not significant. Width, depth, DO and NH_3N explained 49% of the variance. The two habitat parameters alone explained 28% of the variance, whilst the latter two water quality variables explained 35% of the variance. Retaining width and depth, the substitution of DO and NH_3N with the toxicity scores cTOXT1A, cTOXR1A cTOXR3A, cTOXR1T and cTOXR3T explained 53%, 56%, 57%, 52% and 57% respectively. Slope, though significantly correlated with biomass, did not explain variance to any further degree. This variable was negatively intercorrelated with width.

The biomass of this species seemed to be most dependent on the variables used in the analysis of fish species biomass in the Cyprinid region. The species seemed to show a higher biomass in shallow rivers of low width, with high DO concentrations and raised NH_3N level, perhaps indicating a tolerance to organic enrichment. Biomass was positively correlated with toxicity scores, again suggesting raised biomass in shallow organically polluted, but well aerated, rivers.

SECTION 4 - DISCUSSION

The basic approach in this study has been to investigate relationships between fisheries and water quality on a national scale. This has had the effect of losing detailed resolution in the data analysis, but this has perhaps been compensated for by a better overview of nationwide relationships. In addition to gaining a national perspective, individual catchments have also been studied and have yielded valuable results.

Identification of relationships has been hampered by a lack of data from certain regions (for various reasons) and water quality/habitat associations (eg poor water quality in upland areas). Where data has been available, water quality characterisation has generally been poor and fishery data of inconsistent quality. However, the study has shown that relationships can be found at a national level as well as a local level, and this has implications for the standardisation of approaches across pollution control authority boundaries, particularly between NRA regions.

The approaches used in this study have highlighted a number of probable water quality effects in different habitat types, notwithstanding the list of confounding factors discussed in Section 3.2. Of the toxicants represented adequately in the database, total ammonia (un-ionised ammonia was not included in the analysis) appeared to be the greatest influence on fish distributions, although its importance was often obscured by intercorrelations with DO and nitrite. The number of species in Huet's 'Grayling zone' was negatively correlated with mean total ammonia which, along with DO and nitrite, accounted for 49.7% of the variance in species number. The number of fish species was also negatively correlated with mean total ammonia in the Nene catchment, accounting for 72.2% of the variance (although again ammonia was intercorrelated with DO and nitrite). Mean total ammonia accounted for 36.1% of the variance in salmonid biomass in the Perry catchment. In addition to the importance of ammonia and the intercorrelated but lesser effects of DO and nitrite, suspended solids appeared to play a significant role in lowland river stretches, being negatively correlated with coarse fish biomass in Huet's 'Bream zone' and accounting for 33.8% of the variance on its own and 45.8% in combination with total ammonia and DO.

The study has shown the importance of accounting for broad variations in habitat; however, habitat and water quality effects have been shown to be highly intercorrelated, to a degree where zonation into broad habitat types is not sufficient to separate them. It is likely that better correlations between fishery and water quality status would be produced if site-specific habitat diversity (quality) was adequately described.

The failure of association analysis (mentioned in Section 3.3.2) to discern readily-interpretable site groupings based on fish assemblages was mainly due to the intercorrelation of habitat and water quality effects. An alternative approach would involve reforming the database and repeating the association analysis, holding one effect constant whilst assessing the impact of the other. This would either involve studying unimpacted sites with a wide range of habitat types and diversity, or looking at a group of sites with very similar habitat characteristics and a wide range of water quality. The former approach would involve subsequent comparison of impacted sites with reference unimpacted site groups of similar habitat. Both approaches would involve expansion of the existing database in a more structured manner.

Simple univariate analysis of the data followed by MRA has enabled the detection of a number of effects on fisheries, which need to be investigated further. It is unfortunate that more local knowledge of extraneous factors affecting specific sites could not be brought to bear in the time allowed. It should also be noted that no mathematical transformations of the data were attempted, although they may have improved the amount of variance accounted for in the fisheries data.

The toxicity-based approach used in the study has provided useful indications of toxic effect, but has been hampered by the paucity of water quality data available. This approach provides a practical way of assessing the combined effects of toxicants and also has a potential application in consent setting, in the assessment of receiving water quality in relation to envisaged polluting inputs. However, further work is required on the toxicological database.

Although the rule-based approach (using an 'expert' system) has not identified indicator species for differing levels of pollution, it has provided clear indications of its usefulness. The production of sets of rules that environmental managers can consult in order to assess the potential benefit of restocking or enhancing, or to determine which water quality constraints need attention before a healthy fishery can be established is of great potential value. Such an approach provides a framework for a more objective assessment of the constraints acting upon fisheries. It is possible to feed an expert system with ecotoxicological knowledge so that it can judge derived rules for environmental relevance, and this would be an important aspect of any further development work.

SECTION 5 - CONCLUSIONS

In general, the database gathered in this study did not lend itself to the detailed examination of relationships between fishery status and water quality. Water quality information was available for few determinands, usually at a low sampling frequency, and fisheries data was of inconsistent quality.

Notwithstanding the above, a number of probable water quality effects on fishery status and selected fish species have been identified by a variety of statistical techniques, which could be further utilised in the development of fisheries-related EQSs.

Of the parameters recorded at a sufficient frequency in the database, ammonia appeared to have the greatest effect on fish populations, although its importance was often obscured by intercorrelations with DO and nitrite which are also likely to affect fish distribution. Increases in suspended solids coincided with reduced fish biomass in lowland river reaches.

Total fish biomass did not relate to water quality parameters as well as did species number, although the latter was more dependent on habitat type and therefore more obscured in analyses where the habitat range was great.

The use of total fish biomass as an indicator of fishery health has developed from its use in stock management to assess angling potential. It provides no information on the sensitivity or diversity of the species assemblage.

The use of presence/absence information, either on individual species or grouped together as a measure of diversity (species number in its simplest form), takes species sensitivity broadly into account. Qualitative surveys are also less demanding on resources than quantitative surveys, such that a greater number can be performed for the same amount of effort, giving a better geographical coverage. However, species diversity really needs to be related to the number of species which could potentially inhabit a site, unless only sites with very similar habitat characteristics are being compared.

The confounding effect of variation in habitat type on water quality effects was highly significant in the undivided database, and was still evident after its division into broad habitat types. Furthermore, a large proportion of the unexplained variance in fishery status is likely to be due to variation in habitat quality, which could not be quantified in this study. Both observations highlight the need for adequate description of habitat type and quality if the more insidious effects of water quality are to be fully understood.

Relationships between fishery status and NWC class were again obscured by habitat effects, but also by the large within-class variations in fishery parameters. Identification of relationships would have benefited from a more objective and structured approach to site selection; however, some significant differences likely to be due to water quality were evident, particularly within each habitat type.

Temporal variation in fishery status at fixed sites in relation to water quality could not be investigated due to a lack of appropriate data.

The relationship between fishery and invertebrate status was generally poor, although correlations between each of them and water quality characteristics produced coefficients which were normally of the same sign.

The linking of fishery, invertebrate and water quality sites in this study was hampered by a lack of coordination in site location between the relevant groups within most pollution control authority regions. Although it is recognised that the location of biological sites is often influenced by the presence of suitable habitat, if habitat was adequately described and accounted for this would not necessarily pose a serious problem.

Rule-based 'expert' systems show great potential as decision-making aids for environmental managers.

The toxicity approach used in this study could be a valuable tool for detecting general toxic effects and subsequently pin-pointing active toxicants.

SECTION 6 - RECOMMENDATIONS

6.1 GENERAL

Better coordination between the location of fisheries, invertebrate, water quality and, where possible, hydrological monitoring sites is required for a fully integrated assessment of water quality.

A range of standard survey methodologies based on habitat type, in combination with standardised data recording methods, is required in order to ensure full data compatibility and thus maximise the benefits of data collection on a national scale. Greater coordination and dissemination of such standardised information would result in reduced overall effort from individual regions.

Detailed and standardised recording of habitat type and quality are required to better evaluate the confounding physical constraints acting on fish populations. This would facilitate the identification of water quality constraints.

The quinqennial River Quality Survey requires far more stringent controls on acceptable sampling frequency and class assessment protocol than is currently the case. This may improve the relationships found between river quality class and biological (fishery and invertebrate) status.

6.2 FUTURE RESEARCH

A more detailed investigation of the database may reveal further information. This would involve the incorporation of more local knowledge and further use of data transformation. However, the limitations of the database imposed by data compatibility and quality must be borne in mind, and the possibility of conducting field studies for the production of high quality purpose-oriented data should be considered.

If the database were to be studied further, it would need to be expanded to include a more even spread of sites in terms of geography and NWC class/habitat-type associations in order to adequately represent the nationwide scene.

Association analysis, using either a selection of unimpacted sites with a wide variety of habitat type and quality, or a selection of sites with very similar habitat but a wide range of water quality, would provide a useful approach to the detection of disruption within fish communities due to water quality effects.

Temporal variations in fishery status in relation to prevailing water quality and other influencing factors require investigation.

The use of 'expert' systems for the detection of threshold levels of water quality for different species requires further examination. This should focus on the incorporation of ecotoxicological knowledge into the production of rules.

The possibility of formulating a sensitivity index for UK fish species on the basis of further analysis of relationships between fish assemblage and water quality requires investigation. This would also involve a detailed examination of autecological tolerance limits in order to assess the natural range of each species.

Further research is required to identify those fishery parameters which best indicate fishery status and which could be used as a basis for a system of fisheries classification.

Further development of the toxicity-based approach used in this study would involve investigation of the variation in sensitivity between species, and further examination of the influence of water chemistry on the toxicity of certain parameters. Such studies would concentrate on the determination of chronic thresholds rather than acute toxicities.

ACKNOWLEDGEMENTS

We would like to thank all of those people in the NRA, River Purification Boards, the Department of Agriculture and Fisheries for Scotland (DAFS), the Department of Agriculture in N Ireland (DANI) and the Department of the Environment in N Ireland (DoE(NI)) who supplied data for our study, and apologise for cases where information could not be used fully due to compatibility problems. We are particularly grateful to Phil Hickley (NRA Severn Trent Region), David Cragg-Hine (NRA North West Region), Ross Gardiner (DAFS Freshwater Fisheries Laboratory) and Walter Crozier (DANI) for their efforts in collating fisheries data.

REFERENCES

AGG A R and SEAGER J (1989) Progress report on the development of water quality classification schemes. WRc report PRS 2263-M.

ALABASTER J S and LLOYD R (1982) Water Quality Criteria for Freshwater Fish (2nd Ed). Food and Agriculture Organisation of the United Nations. Butterworth Scientific.

ALABASTER J S, GARLAND J H N, HART I C AND SOLBE J F de L G (1972) An approach to the problem of pollution and fisheries. Symp zool Soc Lond 29, 87-114.

ARMITAGE P D, MOSS D, WRIGHT J F and FURSE M T (1983) The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running water sites. Water Res 17(3), 333-347.

ATCHISON G J, HENRY M J and SANDHEINRICH M B (1987) Effects of metals on fish behaviour: a review. Environ Biol Fish 18(1), 11-25.

BALON E K and STEWART D J (1983) Fish assemblages in a river with unusual gradient (Luongo, Africa - Zaire system), reflections on river zonation, and description of another new species. Environ Biol Fish 9, 225-252.

BROWN V M (1968) The calculation of the acute toxicity of mixtures of poisons to rainbow trout. Water Res 2, 723-733.

BROWN V M, GARDINER J and YATES J (1984) Proposed environmental quality standards for List II substances in water: inorganic lead. WRc Technical Report TR 208.

BROWN V M, SHURBEN D G and SHAW D (1970) Studies on water quality and the absence of fish from some polluted English Rivers. Water Res 4, 363-382.

BRUYLANTS B, VANDELANNOOTE A and VERHEYEN R (1986) The movement pattern and density distribution of perch, Perca fluviatilis L, in a channelised lowland river. Aquaculture and Fisheries Management 17, 49-57.

CARPENTER K E (1928) Life in Inland Waters. Sidgwick and Jackson, London.

COOPER V A and SOLBE J F de L G (1978) The fish populations and water quality of the River Tean, Staffordshire. WRc Laboratory Report LR 851.

COUNCIL OF THE EUROPEAN COMMUNITIES (CEC)(1976) Directive of 4 May 1976 on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community. 76/464/EEC; OJL 129.

COUNCIL OF THE EUROPEAN COMMUNITIES (CEC)(1978) Directive of 18 July 1978 on the quality of freshwaters needing protection or improvement in order to support fish life. 78/659/EEC; OJL 222.

COWX I G and BROUGHTON N M (1986) Changes in the species composition of anglers' catches in the River Trent (England) between 1969 and 1984. J Fish Biol 28, 625-636.

DEPARTMENT OF THE ENVIRONMENT (1970) Report of a River Pollution Survey of England and Wales. HMSO.

DEPARTMENT OF THE ENVIRONMENT (1989) Water and the Environment: the implementation of European Community directives on pollution caused by certain dangerous substances discharged into the aquatic environment. DoE circular 7/89.

EDWARDS R W, BENSON-EVANS K, LEARNER M A, WILLIAMS P, WILLIAMS R (1972) A Biological Survey of the River Taff. Wat Pollut Control, Lond 71, 144-166.

EPA (1986) Quality Criteria for Water 1986. United States Environmental Protection Agency, EPA 440/5-86-001. Washington DC.

GARDINER J and MANCE G (1984) United Kingdom water quality standards arising from European Community directives. WRc Technical Report TR 204.

GERKING S D (1953) Evidence for the concepts of home range and territory in stream fishes. Ecology 34, 347-365.

GREENFIELD J P and IRELAND M P (1978) A survey of the macrofauna of a coal-waste polluted Lancashire fluvial system. Environ Pollut 16, 105-122.

HAWKES H A (1975) River zonation and classification. In: River Ecology - Studies in Ecology Volume 2, edited by B A Whitton. Blackwell Scientific.

HELLAWELL J M (1986) Biological indicators of freshwater pollution and environmental management. Pollution monitoring series. Elsevier Applied Science.

HENDERSON P A (1985) An approach to the prediction of temperate freshwater fish communities. J Fish Biol 27 (Supplement A), 279-291.

HERBERT D W M, ALABASTER J S, DART M C and LLOYD R (1961) The effect of china-clay waste on trout streams. Int J Wat Poll 5, 56-74.

HERBERT D W M, JORDAN D H M and LLOYD R (1965) A study of some fishless rivers in the industrial Midlands. J Proc Inst Sew Purif, 569-582.

HOWELLS E J, HOWELLS M E and ALABASTER J S (1983) A field investigation of water quality, fish and invertebrates in the Mawddach river system, Wales. J Fish Biol 22(4), 447-469.

HUET M (1959) Profiles and biology of Western European streams as related to fish management. Trans Am Fish Soc 88(3), 155-163.

HYNES H B N (1960) The Biology of Polluted Waters. Liverpool University Press.

HYNES H B N (1979) The Ecology of Running Waters. 4th edition. Liverpool University Press.

JORDAN D R and WORTLEY J S (1985) Sampling strategy related to fish distribution, with particular reference to the Norfolk Broads. J Fish Biol 27 (Supplement A), 163-173.

KARR J R (1981) Assessment of biotic integrity using fish communities. Fisheries (Bethesda) 6(6), 21-27.

KAY J, BROWN M W, CRYER A, SOLBE J F de L G, SHURBEN, GARVEY J S and THOMAS D G (1987) Metallothionein gene expression and cadmium toxicity in freshwater fish. Experientia Supplementum 52, 627-630.

KLINE E R, MALTSON V R, PICKERING Q H, SPEHAR D L, STEPHAN C E (1987) Effects of pollution on freshwater organisms. J Water Poll Con Fed 59(6), 539-572.

KUEHNE R A (1962) A classification of streams, illustrated by fish distribution in an eastern Kentucky creek. Ecology 43, 608-614.

LEARNER M A, WILLIAMS R, HARCUP M and HUGHES B D (1971) A survey of the macro-fauna of the River Cynon, a polluted tributary of the River Taff (South Wales). Freshwater Biol 1, 336-367.

LELEK A (1981) Population dynamics of fishes in the changing streams. In: Dynamique de Populations et Qualite de L'eau, edited by H Hoestlandt. Gauthier-villars, Paris.

LETTERMAN R D and MITSCH W J (1978) Impact of mine drainage on a mountain stream in Pennsylvania. Environ Pollut 17, 53-73.

LINFIELD R S J (1985) An alternative concept to home range theory with respect to populations of cyprinids in major river systems. J Fish Biol 27 (Supplement A), 187-196.

MANCE G and CAMPBELL J A (1988) Proposed environmental quality standards for List II substances in water: iron. WRc Technical Report TR 258.

MANCE G and YATES J (1984a) Proposed environmental quality standards for List II substances in water: zinc. WRc Technical Report TR 209.

MANCE G and YATES J (1984b) Proposed environmental quality standards for List II substances in water: nickel. WRc Technical Report TR 211.

MANCE G, BROWN V M, GARDINER J and YATES J (1984a) Proposed environmental quality standards for List II substances in water: chromium. WRc Technical Report TR 207.

MANCE G, BROWN V M and YATES J (1984b) Proposed environmental quality standards for List II substances in water: copper. WRc Technical Report TR 210.

MANCE G, MUSSELWHITE C and BROWN V M (1984c) Proposed environmental quality standards for List II substances in water: arsenic. WRc Technical Report TR 212.

MANCE G, NORTON R and O'DONNELL A R (1988a) Proposed environmental quality standards for List II substances in water: vanadium. WRc Technical Report TR 253.

MANCE G, O'DONNELL A R, CAMPBELL J A and GUNN A M (1988b) Proposed environmental quality standards for List II substances inwater: inorganic tin. WRc Technical Report TR 254.

MANCE G, O'DONNELL A R and SMITH P R (1988c) Proposed environmental quality standards for List II substances in water: boron. WRc Technical Report TR 256.

MANCE G, O'DONNELL A R and CAMPBELL J A (1988d) Proposed environmental quality standards for List II substances in water: sulphide. WRc Technical Report TR 257.

MATTHEWS W J (1986) Fish faunal 'breaks' and stream order in the eastern and central United States. Environ Biol Fish 17(2), 81-92.

MAYER F L and ELLERSIECK M R (1986) Manual of aquatic toxicity: Interpretation and database for 410 chemicals and 66 species of freshwater animals. United States Dept of the Interior Fish and Wildlife Service, Resource Publication 160. Washington DC.

MILNER N J, SCULLION J, CARLING P A, CRISP D T (1981) The effects of discharge on sediment dynamics and consequent effects on invertebrates and salmonids in upland rivers. Advances in Applied Biology 4, 153-220.

MILNER N J, HEMSWORTH R J and JONES B E (1985) Habitat evaluation as a fisheries management tool. J Fish Biol 27 (Supplement A), 85-108.

NEWMAN R M and WATERS T F (1989) Differences in brown trout production among contiguous sections of an entire stream. Can J Fish Aquat Sci 46, 203-213.

O'DONNELL A R, MANCE G AND NORTON R (1984) A review of the toxicity of aluminium in freshwater. WRc Technical Report TR 197.

OVERREIN L N, SEIP H M and TOLLAN A (1981) Acid Precipitation-effects on forest and fish. Final Report of the SNSF-project 1972-1980. 2nd edition, Oslo.

PERSOONE G (1979) Proposal for a biotypological classification of watercourses in the European Communities. In: Biological Indicators of Water Quality, edited by A James and L Evison. John Wiley & Sons.

PICKERING Q, CARLE D O, PILLI A, WILLINGHAM T, LAZORCHAK J M, (1989) Effects of pollution on freshwater organisms. J Water Poll Con Fed 61(6), 998-1042.

PIHAN J C and LANDRAGIN G (1985) Modele de prevision de la toxicite aigue des eaux de rivieres pour le poisson. Verh Internat Verein Limnol 22, 2457-2462.

PITWELL L R (1976) Biological monitoring of rivers in the Community. In: Principle and methods for determining ecological criteria on hydrobiocenoses. Proceedings of the European Scientific Colloquium, Luxembourg 1975.

PORTT C B, BALON E K, and NOAKES D L (1986) Biomass and production of fishes in natural and channelised streams. Can J Fish Aq Sci **43**, 1926-1934.

SCHOFIELD C L and DRISCOLL C T (1987) Fish distribution in relation to water quality gradients in the North Branch of the Moose River Basin. Biogeochem 3, 63-85.

SCULLION J and EDWARDS R W (1980a) The effects of coal industry pollutants on the macroinvertebrate fauna of a small river in the South Wales coalfield. Freshwater Biol **10**, 141-162.

SCULLION J and EDWARDS R W (1980b) The effects of pollutants from the coal industry on the fish fauna of a small river in the South Wales coalfield. Environ Pollut (Series A) **21**, 141-153.

SEAGER J, WOLFF E W and COOPER V A (1988) Proposed environmental quality standards for List II substances in water: ammonia. WRc Technical Report TR 260.

SEBER G A F and LE CREN E D (1967) Estimating population parameters from catches large relative to the population. J Anim Ecol **36**, 631-643.

SMYLY W J P (1957) The life-history of the bullhead or miller's thumb (Cottus gobio L). Proc Zool Soc Lond **28**, 431-453.

SOLBE J F de L G (1973) The relationship between water quality and the status of fish populations in Willow Brook. Proc Soc Water Treat Exam 22, 41-61.

SOLBE J F de L G (1977) Water quality, fish and invertebrates in a zinc-polluted stream. In: Biological Monitoring of Inland Fisheries, edited by J S Alabaster.

SOLBE J F de L G and COOPER V A (1975) Progress report on study of relations between water quality and fish populations in the River Churnet, N Staffs. WRc Technical Memorandum TM 113.

SOLBE J F de L G and FLOOK V A (1975) Studies on the toxicity of zinc sulphate and of cadmium sulphate to stone loach, Noemacheilus barbatulus (L.), in hard water. J Fish Biol 7, 631-637.

SPEHAR R L, CARLSON R W, LEMKE A E, MOUNT D I, PICKERING Q H and SNARSKI V M (1980) Effects of pollution on freshwater fish. J Water Poll Con Fed 52(6), 1703-1768.

STAUFFER J R, HOCUTT C H, MASNIK M T, and REED J E (1975) The longitudinal distribution of the fishes of the East River, West Virginia - Virginia. Virginia Journal of Science 26, 121-125.

STONER J H, GEE A S and WADE K R (1984). The effects of acidification on the ecology of streams in the upper Tywi catchment in West Wales. Environ Pollut (Series A) 35, 125-157.

STOTT B (1967) The movements and population densities of roach (Rutilus rutilus L) and gudgeon (Gobio gobio L) in the River Mole. J Anim Ecol 36, 407-423.

SWALES S (1982) A 'before and after' study of the effects of land drainage works on fish stocks in the upper reaches of a lowland river. Fish Mgmt 13, 105-115.

SWALES S (1988) Fish populations of a small lowland channelised river in England subject to long-term maintenance and management works. Regulated rivers: research and management 2, 493-506.

TAYLOR B R and ROFF J C (1986) Long-term effects of highway construction on the ecology of a Southern Ontario stream. Environ Pollut (Series A) 40, 317-344.

TOMBLESON P H (1974) Chairman: Coarse Fisheries. A Report of the Working Party of the Association of River Authorities.

TSAI C-F (1970) Changes in fish populations and migration in relation to increased sewage pollution in Little Patuxent River, Maryland. Chesapeake Science 11(1), 34-41.

TSAI C-F (1971) Water Quality Criteria to Protect the Fish Population Directly below Sewage Outfalls. Dept Forestry, Fish and Wildlife Natural Resources Institute, Univ of Maryland.

TURNPENNY A W H, SADLER K, ASTON R J, MILNER A G P and LYNAM S (1987) The fish populations of some streams in Wales and northern England in relation to acidity and associated factors. J Fish Biol 31, 415-434.

US FISH & WILDLIFE SERVICE (1980) Habitat as a basis for environmental assessment. Fourth release of the Ecological Services Manual. Transmittal Sheet 101 ESM. Department of the Interior, Washington DC.

VAN LOON J C and BEAMISH R J (1977) Heavy-metal contamination by atmospheric fallout of several Flin Flon area lakes and the relation to fish populations. J Fish Res Board Can 34, 899-906.

WELSH WATER (1986) Regional Acid Waters Electrofishing Survey Interim Report. Scientific Services South Western District Report No SW/86/19.

WHEELER A (1974) Changes in the freshwater fish fauna of Britain. In: The Changing Flora and Fauna of Britain, edited by Hawkesworth. Systematics Association Special Volume. Academic Press.

WILLIAMS R and HARCUP M F (1974) The fish populations of an industrial river in South Wales. J Fish Biol 6, 395-414.

WOLFF E W, SEAGER J, COOPER V A AND ORR J (1988) Proposed environmental quality standards for List II substances in water: pH. WRc Technical Report TR 259.

YOUNG S N (1985) Fishery Ammonia EQS Evaluation - statistical analysis of Anglian Water data. Anglian Water.

ZABEL T F, SEAGER J, OAKLEY S D (1988a) Proposed environmental quality standards for List II substances in water: organotins. WRc Technical Report TR 255.

ZABEL T F, SEAGER J, OAKLEY S D (1988b) Proposed environmental quality standards for List II substances in water: mothproofing agents. WRc Technical Report TR 261.

ZALEWSKI M and NAIMAN R J (1985) The regulation of riverine fish communities by a continuum of abiotic-biotic factors. In: Habitat Modifications and Freshwater Fisheries, edited by J S Alabaster. Butterworth, London.

ZALEWSKI M, FRANKIEWICZ P, and BREWINSKA B (1985) The factors limiting growth and survival of brown trout, Salmo trutta m fario L, introduced to different types of streams. J Fish Biol 27 (Supplement A), 59-73.

ZIPPIN C (1958) The removal method of population estimation. J Wildlife Management 22, 82-90.

APPENDIX A - ENVIRONMENTAL STANDARDS CONCERNING THE PROTECTION OF FRESHWATER FISH

APPENDIX A.1. Existing and proposed UK Environmental Quality Standards relating to the protection of freshwater fish and freshwater life in general.

(For further explanation refer to the sources indicated)

General physico-chemical parameters

PARAMETER	WATERS COVERED	VALUE	UNITS (µg/l unless otherwise stated)	SOURCE
NH3	EC designated salmonid waters	0.031 TG / 0.78 TI	mg/l annual average	Gardiner & Mance 1984
		4.12 UG / 20.6 UI	annual average	Gardiner & Mance 1984
	EC designated cyprinid waters	0.16 TG / 0.78 TI	mg/l annual average	Gardiner & Mance 1984
		4.12 UG / 20.6 UI	annual average	Gardiner & Mance 1984
(proposed)	EC designated waters (salmonid & cyprinid waters)	0.021 UI / 0.78 TI	mg/l 95% compliance	Seager et al 1988
	non-EC designated waters	0.015 U	mg/l annual average	Seager et al 1988
Diss. Oxygen	EC designated salmonid waters	9 I	mg/l 50% compliance	Gardiner & Mance 1984
		7 G	mg/l 100% compliance	Gardiner & Mance 1984
		6	mg/l remedial action required	Gardiner & Mance 1984
	EC designated cyprinid waters	7 I	mg/l 50% compliance	Gardiner & Mance 1984
		5 G	mg/l 100% compliance	Gardiner & Mance 1984
		4	mg/l remedial action required	Gardiner & Mance 1984
BOD	EC designated salmonid waters	3 G	mg/l annual average	Gardiner & Mance 1984
	EC designated cyprinid waters	6 G	mg/l annual average	Gardiner & Mance 1984
Nitrite	EC designated salmonid waters	3 G	annual average	Gardiner & Mance 1984
	EC designated cyprinid waters	9 G	annual average	Gardiner & Mance 1984
Temperature	EC designated salmonid waters	21.5	°C 98% compliance	Gardiner & Mance 1984
	EC designated cyprinid waters	28	°C 98% compliance	Gardiner & Mance 1984
Susp. solids	EC designated sal & cyp waters	25 G	mg/l annual average	Gardiner & Mance 1984
PO4	EC designated salmonid waters	65 G	annual average	Gardiner & Mance 1984
	EC designated cyprinid waters	131 G	annual average	Gardiner & Mance 1984
Residual Cl2	EC designated waters	6.8 I	annual average	Gardiner & Mance 1984
pH	Sensitive aquatic life (eg salmonid fish)	6.0 - 9.0 I	95% compliance	DoE 1989
	Other aquatic life (eg cyprinid fish)	6.0 - 9.0	95% compliance	DoE 1989
(proposed)	Other freshwater life	6.5 - 8.5	annual average	Wolff et al 1988

U = un-ionised NH3
T = total NH3
G = Guide value
I = Mandatory value

APPENDIX A.1 (cont.)

Dangerous substances under EC Directive 76/464/EEC

LIST I

PARAMETER	WATERS COVERED	VALUE	UNITS ($\mu g/l$ unless otherwise stated)	SOURCE
Mercury	Inland surface waters	1	annual average (total)	DoE 1989
Cadmium	Inland surface waters	5	annual average (total)	DoE 1989
Hexachloro-cyclohexane (HCH)	Inland surface waters	0.1	annual average (total)	DoE 1989
CCl4	All waters	12	annual average	DoE 1989
DDT	All waters	0.025	annual average	DoE 1989
Pentachlorophenol	All waters	2	annual average	DoE 1989
The 'drins'	All waters	0.03	annual average (total)	DoE 1989
Hexachlorobenzene	All waters	0.03	annual average	DoE 1989
Hexachlorobutadiene	All waters	0.1	annual average	DoE 1989
Chloroform	All waters	12	annual average	DoE 1989

APPENDIX A.1 (cont.)

Dangerous substances under EC directive 76/464/EEC (cont.)

LIST II

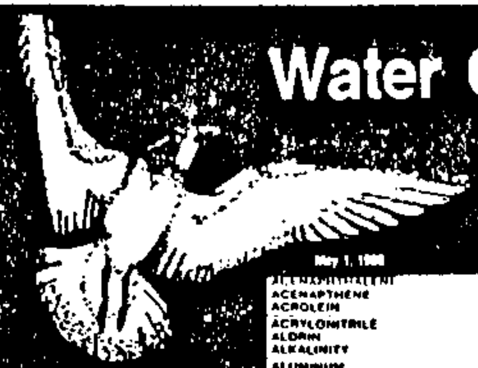
PARAMETER	WATERS COVERED	VALUE	UNITS ($\mu g/l$ unless otherwise stated)	SOURCE
Lead	Sensitive aquatic life	4 - 20	annual average (diss.)	DoE 1989
	Other aquatic life	50 - 250	annual average (diss.)	DoE 1989
Chromium	Sensitive aquatic life	5 - 50	annual average (diss.)	DoE 1989
	Other aquatic life	150 - 250	annual average (diss.)	DoE 1989
Zinc	Sensitive aquatic life	8 - 125	annual average (total)	DoE 1989
		30 - 300	95% compliance (diss.)	DoE 1989
	Other aquatic life	75 - 500	annual average (total)	DoE 1989
		300 - 2000	95% compliance (diss.)	DoE 1989
Copper	Sensitive & other aquatic life	1 - 28	annual average (diss.)	DoE 1989
		5 - 112	95% compliance (diss.)	DoE 1989
Nickel	Sensitive & other aquatic life	50 - 200	annual average (diss.)	DoE 1989
Arsenic	Sensitive & other aquatic life	50	annual average (diss.)	DoE 1989
Iron	Sensitive & other aquatic life	1000	annual average (diss.)	DoE 1989
Inorg. Tin (proposed)	Protection of freshwater fish & other FW life	25	average concentration	Mance et al 1988b
Sulphide (proposed)	Freshwater fish	0.25 - 1.0	annual average (as H2S)	Mance et al 1988d
		2.5 - 10.0	24 hr maximum average	Mance et al 1988d
	Other freshwater life	1.0 - 2.0	annual average (as H2S)	Mance et al 1988d
		10.0 - 20.0	24hr maximum average	Mance et al 1988d
Boron	Sensitive & other aquatic life	2000	annual average (total)	DoE 1989
Vanadium	Sensitive & other aquatic life	20 - 60	annual average (total)	DoE 1989
Tributyltin	Sensitive & other aquatic life	0.02	maximum allowable concentration (total)	DoE 1989
Triphenyltin	Sensitive & other aquatic life	0.02	maximum allowable concentration (total)	DoE 1989
Mothproofing agents:				
PCSDs	Sensitive & other aquatic life	0.05	95% compliance (total)	DoE 1989
Cyfluthrin	Sensitive & other aquatic life	0.001	95% compliance (total)	DoE 1989
Sulcofuron	Sensitive & other aquatic life	25	95% compliance (total)	DoE 1989
Flucofuron	Sensitive & other aquatic life	1.0	95% compliance (total)	DoE 1989
Permethrin	Sensitive & other aquatic life	0.01	95% compliance (total)	DoE 1989

Where a range of values is indicated the EQS varies with water hardness (low hardness - low EQS) EXCEPT for sulphide, where the proposed EQS varies with temperature and dissolved oxygen.

NB EQSs for the following substances have been proposed by the Water Research Centre (WRc) to the Department of the Environment but have not yet been adopted: atrazine, simazine, malathion, fenitrothion, dichlorvos, azinphos-methyl, xylenes, toluene, trichlorobenzene, dichlorobenzene, monochlorobenzene. Substances currently under investigation by WRc are: benzene, 1,2-dichloroethane and endosulfan. In addition, EQSs for the List II metals lead, chromium, copper, nickel, zinc and arsenic are under review.

APPENDIX A.2 US Environmental Protection Agency quality criteria for water.

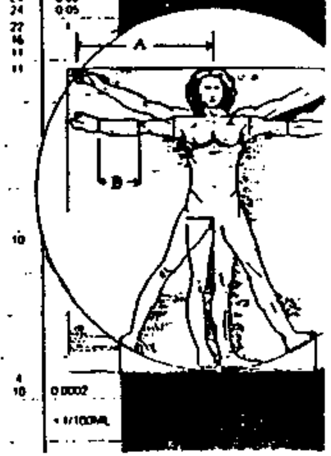
Water Quality Criteria Summary



May 1, 1986

CONCENTRATIONS IN µg/l — HUMAN HEALTH

	Priority Pollutant	Fresh Acute Criteria *LOEL	Fresh Chronic Criteria *LOEL	Marine Acute Criteria *LOEL	Marine Chronic Criteria *LOEL	Water and Fish Ingestion	Fish Consumption Only	Carcinogen	Date Reference	Criteria Document Development	No. of States with Standard AQ LI	HU H	Drinking Water M.C.L.
ACENAPHTHALENE	N												
ACENAPHTHENE	Y	*1 700	*520	*970	*500	20µg			1980 FR	F			
ACROLEIN	Y	*68	*21	*55		320µg	780µg		1980 FR	F			
ACRYLONITRILE	Y	*7 550	*2 600			0.058µg	0.65µg	Y	1980 FR	F			
ALDRIN	Y	3		1.3		0.074ng	0.079ng	Y	1980 FR	F	10	6	
ALKALINITY	N		20 000						1976 RB	F			
ALUMINUM	N									9/30/86			
AMMONIA TOTAL	N	15.7	3.9						1985 FR	F	23	20	
AMMONIA UN-IONIZED	N	0.092	0.022						1985 FR	F	23	20	
ANALINE	N									9/30/86			
ANTIMONY	Y	*9 000	*1 600			146µg	45 000µg		1980 FR	F			
ARSENIC	N	360	190	36	69	2.2ng	17.5ng	Y	1985 FR	F			0.05
ARSENIC(PENT)	N	*850	*48	*2 319	*13			Y	1985 FR	F	22	20	
ARSENIC(TRI)	Y	[illegible]	[illegible]	69	36			Y	1985 FR	F	22	20	
ASBESTOS	Y					30k f/l		Y	1980 FR	F			
BARIUM	N					1mg			1976 RB	F	13	25	1
BENZENE	Y	*5 300		*5 100	*700	0.66µg	40µg	Y	1980 FR	F			
BENZIDINE	Y	*2 500				0.12ng	0.53ng	Y	1980 FR	F	3	4	
BERYLLIUM	Y	*130	*5.3			3.7ng	64.1ng	Y	1980 FR	F	7	4	
BHC	Y	*100		*0.34					1980 FR	F			
BIS 2 CHLOROETHYOXY METHANE	N												
BORON	N								1976 RB	F	5	3	
BUTYLBENZYLPHTHALATE	N								1980 RB				
CADMIUM	Y	3.9	1.1	43	9.3	10µg			1985 FR	F	22	27	0.01
CARBAZOLE	N												
CARBON TETRACHLORIDE	Y	*35 200		*50 000		0.4µg	6.94µg	Y	1980 FR	F			
CHLORALKYL ETHERS	Y	*238 000				0.0038ng	1.84ng	Y	1980 FR				
CHLORDANE	Y	2.4	0.0043	0.004	0.09	0.46ng	0.48ng	Y	1980 FR	F			
CHLORINATED BENZENES	Y	*250	*50	*160	*129				1980 FR	F			
CHLORINATED ETHANES	Y							S	1980 FR	F			
CHLORINATED ETHERS	N												
CHLORINATED NAPHTHALENES	Y	*1 600		*7.5					1980 FR	F			
CHLORINATED PHENOLS	Y	*500 600	970	*440					1980 FR	F			
CHLORINE	N	19	11	13	7.5				1985 FR	F			
CHLORO-4 METHYL-3 PHENOL	N	*30											
CHLOROFORM	Y	*28 900	*1 240			0.19µg	15.7µg	Y	1980 FR	F			
CHLOROPHENOL	N			*440									
CHLOROPHENOL 2	Y	4 380	*2 000				0.1µg		1980 FR	F			
CHLOROPHENOL 4	N			*29 700									
CHLOROPHENOXY HERBICIDES	N								1976 RB	F			
CHROMIUM (HEX)	Y	16	11	1 100	50	50µg			1985 FR	F	2	24	0.05
CHROMIUM (TRI)	Y	1 700	210	10 300					1985 FR	F	2	24	0.05
COPPER	Y	18	12	2.9		170mg	3 433mg		1985 FR	F	20	22	1
CYANIDE	Y	22	5.2	1		200µg			1985 FR	F	19	16	
DDE	Y	*1 050		*14							10	11	
DDT	Y	1.1	0.001	0.13	0.001	0.024ng	0.024ng	Y	1980 FR	F	10	11	
DEHP	N									9/30/86			
DEMETON	N		0.1		0.1				1976 RB	F			
DIAMINO TOLUENE	N												
DIBENZOFURANE	N									9-30-86			
DICHLORINATED ETHANES	N	*11,800	*20 000	*113 000									
DICHLOROBENZENES	Y	*1 120	*763	*1 970		400µg	2.6mg		1980 FR	F			
DICHLOROBENZIDINES	Y					0.0103µg	0.0204µg	Y	1980 FR	F			
DICHLOROETHYLENE	Y	*11 600		*224 000		0.033µg	1.85µg	Y	1980 FR	F			
DICHLOROPHENOL 2,4	N	*2 020	*365				0.3µg		1980 FR	F			
DICHLOROPROPANE	Y	*23 000	*5 700	*10 300	*3 040				1980 FR	F			
DICHLOROPROPENE	Y	*6 060	*244	*790		87µg	14.1mg		1980 FR	F			
DIELDRIN	Y	2.5	0.0019	0.71	0.0019	0.071ng	0.076ng	Y	1980 FR	F	12	10	
DIMETHYL PHENOL	Y								1980 FR	F			
DIMETHYL PHENOL 2,4	Y	*2 120					400		1980 FR	F			
DIMETHYLSULFOXIDE	N												
DINITROBENZENE 1,3	N												
DINITROTOLUENE 2,4	Y	*330	*230	*590	*370	0.11µg	9.1µg	Y	1980 FR	F			
DIOXANE P	N												
DIOXIN	Y	*0.01	*0.0056					Y	1984 FR	F			
DIPHENYLHYDRAZINE	Y	*270			Y	0.42µg	0.56µg		1980 FR	F			
DISSOLVED OXYGEN	N	6 500	4 000						1986 FR	F			
ENDOSULFAN	Y	0.22	0.056	0.034	0.0087	74µg	159µg		1980 FR	F	6	4	
ENDRIN	Y	0.18	0.0023	0.037	0.0023	1µg			1980 FR	F	13	10	0.0002
ETHYLBENZENE	Y	*32 000		*430		1.4mg	3.28mg		1980 FR	F			
FECAL COLIFORM	N												<1/100ML
FLUORANTHENE	Y	*3 980		*40	*16	42µg	54µg		1980 FR	F			



APPENDIX A.2 (cont.)

DISSOLVED OXYGEN	N	8 500	4 000						1986 FR	F			
ENDOSULFAN	Y	0.22	0.056	0.034	0.0087	74µg	159µg		1980 FR	F	8	4	
ENDRIN	Y	0.18	0.0023	0.037	0.0023	1µg			1980 FR	F	13	10	0.0002
ETHYLBENZENE	Y	*32 000		*430		1.4mg	3.28mg		1980 FR	F			
FECAL COLIFORM	N												1/100ML
FLUORANTHENE	Y	*3 980		*40	*16	42µg	54µg		1980 FR	F			
FLUORIDE	N												1.4-2.4
GUTHION	N		*0.01		*0.01				1976 RB	F			
HALOETHERS	Y	*360	*122						1980 FR	F			
HALOMETHANES	Y	*11 000		*12 000	*6 400	0.19µg	15.7µg	Y	1980 FR	F			
HARDNESS	N												
HEPTACHLOR	Y	0.52	0.0038	0.053	0.0036	0.28ng	0.29ng	Y	1980 FR	F	5	3	
HEXACHLOROBENZENE	Y	250	30	160	129								
HEXACHLOROBUTADIENE	Y	*90	*9.3	*32		0.45µg	50µg	Y	1980 FR	F			
HEXACHLOROCYCLOHEXANE (LINDANE)	Y	2	0.08	0.16		9.2ng	31ng	Y	1980 FR	F	7	11	0.004
HEXACHLOROCYCLOPENTADIENE	Y	*7	*5.2	*7		260µg			1980 FR	F			
HEXACHLOROETHYLENES	N												
HEXACHLORINATED ETHANES	Y	*980	*540	*940		1.9µg	8.74µg	Y	1980 FR	F			
IRON	N	*1 000				0.3mg			1976 RB	F	16	17	0.3
ISOPHORONE	Y	*117 000		*12 900		5.2mg	520mg		1980 FR	F			
LEAD	Y	82	3.2	140	5.6	50µg			1985 FR	F	25	24	0.05
MALATHION	N		*0.1		*0.1				1976 RB	F			
MANGANESE	N					50µg	100µg		1976 RB	F	9	11	0.05
MERCURY	Y	2.4	0.012	2.1	0.025	144ng	146ng		1985 FR	F	20	21	0.002
METHOXYCHLOR	N	0.012	*0.03		*0.03	100µg			1976 RB	F			0.1
MIREX	N		*0.001		*0.001				1976 RB	F			
NAPHTHALENE	Y	*2 300	*620	*2 350					1985 FR	F			
NICKEL	Y	*1 800	*96	140	7.1	13.4µg	100µg		1980 FR	F	11	7	
NITRATES	N					10mg			1976 RB	F	20	25	10
NITROANILINE	N												
NITROBENZENE	Y	*27 000		*6 680		30µg			1980 FR	F			
NITROPHENOLS	Y	*230	*150	*4 850		13.4µg	765µg		1980 FR	F			
NITROSAMINES	Y	*5 850		*3 300 000		0.8ng	1 240ng	Y	1980 FR	F			
OIL AND GREASE	N								1976 RB	F			
ORGANOTINS	N									9-30-86			
PARATHION	N		*0.04		*0.04				1976 RB	F			
PCB's	Y	2	0.014	10	0.03	0.079ng	0.79ng	Y	1980 FR	F	11	7	
PENTACHLORINATED ETHANES	Y	*7 240	*1 100	*390	*281								
PENTACHLOROPHENOL	Y	*55	*3.2	*53	*34	1.01mg			1980 FR	F			
PH	N		*6.5-9		*6.5-8.5	5-9			1976 RB	F			
PHENOL	Y	*10 200	*2 560	*5 800		30µg			1980 FR	F	19	20	
PHOSPHORUS	N				*0.01				1976 RB	F	25	19	
PHTHALATE ESTERS	Y	*940	*3	*2 944	3.4	15mg	50mg		1980 FR	F	3	5	
POLYCHLORINATED DIPHENYL ETHERS	N												
POLYNUCLEAR AROMATIC HYDROCARBONS	Y			*300		2.8ng	31.1ng	Y	1980 FR	F			
SALINITY	N												
SELENIUM	Y	260	35	410	54	10µg			1980 FR	F	23	21	0.01
SILVER	Y	4.1	0.12	2.3		50µg			1980 FR	F	14	18	0.05
SOLIDS DISSOLVED	N					250mg			1976 RB	F	58	58	
SOLIDS SUSPENDED	N								1976 RB	F	56	58	
STYRENE	N												
SULFIDE-HYDROGEN SULFIDE	N		*2		*2				1976 RB	F			
TAINTING SUBSTANCES	N								1976 RB	F			
TDE	Y	*0.6		*3.6									
TEMPERATURE	N									F			
TETRACHLORINATED ETHANES	Y	*9 320	*2 400	*9 020					1980 FR				
TETRACHLOROBENZENE 1,2,4,5	Y	*250	*50	*160	*129	38µg	48µg		1980 FR	F			
TETRACHLOROETHYLENE	Y	*5 280	*840	*10 200	*450	0.8µg	8.85µg	Y	1980 FR	F			
TETRACHLOROPHENOL 2,3,4,6	Y			440		1µg			1980 FR	F			
TETRAMETHYL LEAD	N												
THALLIUM	Y	*1 400	*40	*2 13		13µg	48µg		1980 FR	F			
TOLUENE	Y	*17 500		*6 300	*5 000	14.3mg	424mg		1980 FR	F			
TOXAPHENE	Y	1.6	0.013	0.07		0.71ng	0.73ng	Y	1980 FR	F			0.005
TRICHLORINATED ETHANES	Y	*18 000	*9 400	*31 200		0.6µg	41.8µg	Y	1980 FR	F			
TRICHLOROBENZENE	Y								1980 FR	F			
TRICHLOROBENZENE 1,2,3	Y								1980 FR	F			
TRICHLOROETHYLENE	Y	*45 000	*21,900	*2 000		2.7µg	80.7µg	Y	1980 FR	F			
TRICHLOROPHENOL 2,3,6	Y												
TRICHLOROPHENOL 2,4,5	Y					1µg			1980 FR	F			
TRICHLOROPHENOL 2,4,6	N		*970			1.2µg	3.6µg	Y	1980 FR	F			
VINYL CHLORIDE	Y					2µg	525µg	Y	1980 FR	F			
XYLENE	N												
ZINC	Y	320	47	*170	58	5mg			1980 FR	F	17	18	5

* WATER QUALITY DEPENDENT CRITERION. SEE DOCUMENT.

mg milligrams per liter
ng nanograms per liter
µg micrograms per liter

Y YES
N NO

* LOEL LOWEST OBSERVED EFFECT LEVEL

Y YES
S SUSPECT

FR FEDERAL REGISTER
RD RED BOOK

F FINAL
M/D/Y EXPECTED DATE OF FINAL DOCUMENT

CHANGES, CORRECTIONS AND SUGGESTIONS SHOULD BE ADDRESSED TO
CHRISTOPHER S. ZARBA
U.S. ENVIRONMENTAL PROTECTION AGENCY
401 M STREET S.W. (WH-585)
WASHINGTON DC 20460

APPENDIX B - DETAILS OF STUDY SITES

APPENDIX B. DETAILS OF STUDY SITES

(Dates indicate the time of surveying - fisheries and invertebrate sites only)

NRA ANGLIAN REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
AN1	NENE	DITCHFORD LOCK CHANNEL DATE 20/10/83 NGR SP 928 683	DITCHFORD MILL DATE 25/7/83 NGR SP 931 684	DITCHFORD MILL LOCK CHANNEL NRA CODE RO5BFNENE260D NGR SP 930 682
AN2	NENE	IRTHLINGBOROUGH DATE 21/10/83 NGR SP 957 703	IRTHLINGBOROUGH DATE 20/7/82 NGR SP 956 705	IRTHLINGBOROUGH OLD RD BR NRA CODE RO5BFNENE300I NGR SP 957 706
AN3	NENE	RINGSTEAD LOCK CHANNEL DATE 25/10/83 NGR SP 974 752	UPPER RINGSTEAD LOCK DATE 28/7/83 NGR SP 967 746	RINGSTEAD ROAD BRIDGE NRA CODE RO5BFNENE340R NGR SP 974 752
AN4		REJECTED		
AN5	NENE	OUNDLE DATE 1/12/83 NGR -	---	OUNDLE (NORTH) RD BR NRA CODE RO5BFNENE460O NGR TL 045 889
AN6	NENE	FOTHERINGHAY DATE 19/6/84 NGR TL 059 929	---	FOTHERINGHAY ROAD BRIDGE NRA CODE RO5BFNENE490F NGR TL 061 929
AN7	NENE	WANSFORD DATE 26/6/84 NGR TL 081 996	WANSFORD DATE 1/8/83 NGR TL 080 995	WANSFORD OLD RD BR NRA CODE RO5BFNENE550W NGR TL 075 991
AN8	WILLOW BROOK (NENE)	SOUTHERN STREAM WELDON DATE 28/11/84 NGR SP 928 894	---	SOUTHERN TRIB A427 GT WELDON NRA CODE RO5BFWILL030S NGR 930 895
AN9	WILLOW BROOK (NENE)	CENTRAL STREAM WELDON DATE 28/11/84 NGR SP 921 897	CHURCH ROAD WELDON CENTRAL ST DATE 12/12/84 NGR SP 929 894	CENTRAL TRIB WATER LANE WELDON NRA CODE RO5BFWILL020C NGR SP 921 896
AN10		REJECTED		
AN11	WILLOW BROOK (NENE)	NORTHERN STR WELDON LODGE DATE 4/12/84 NGR SP 917 915	WELDON LODGE NORTHERN STREAM DATE 9/10/84 NGR SP 917 915	NORTHERN TRIB WELDON LODGE NRA CODE RO5BFWILL010N NGR SP 917 915
AN12	WILLOW BROOK (NENE)	D/S DEENE LAKE DATE 30/11/84 NGR SP 955 929	OUTFLOW FROM DEENE LAKE DATE 12/12/84 NGR SP 955 927	DEENE LAKE ROAD BRIDGE NRA CODE RO5BFWILL040D NGR SP 955 928
AN13	WILLOW BROOK (NENE)	BULWICK D/S BRIDGE DATE 30/11/84 NGR SP 963 945	BULWICK MILL DATE 5/10/83 NGR SP 962 943	BULWICK A43 RD BR NRA CODE RO5BFWILL050B NGR SP 962 943
AN14	WILLOW BROOK (NENE)	KINGS CLIFFE (FISHPOOL MEADOW) DATE 11/12/84 NGR TL 001 969	KINGSCLIFFE DATE 6/10/83 NGR TL 007 971	KINGS CLIFFE ROAD BRIDGE NRA CODE RO5BFWILL110K NGR TL 009 971

APPENDIX B (cont.)

AN15	WILLOW BROOK (NENE)	FOTHERINGHAY DATE 13/12/84 NGR TL 060 934	WB FOTHERINGHAY DATE 16/7/84 NGR TL 062 935	WB FOTHERINGHAY ROAD BRIDGE NRA CODE R05BFWILL170F NGR TL 063 935
AN16	BLACKWATER	LYONS HALL DATE 20/2/87 NGR TL 777 243	---	@ STRAITS MILL NRA CODE R01BFBL0675 NGR TL 770 243
AN17	BLACKWATER	COVENBROOK HALL DATE 7/5/87 NGR TL 788 248	---	@ STISTED MILL NRA CODE R01BFBL06 NGR TL 790 246
AN18	BLACKWATER	BRADWELL BRIDGE DATE 6/5/87 NGR TL 805 234	---	@ BRADWELL BRIDGE NRA CODE R01BFBL05 NGR TL 807 231
AN19	BLACKWATER	POINTWELL MILL DATE 31/3/87 NGR TL 854 217	---	@ POINTWELL MILL NRA CODE R01BFBL0417 NGR TL 853 215
AN20	BLACKWATER	FEERINGBURY HALL DATE 1/4/87 NGR TL 863 214	FEERINGBURY FARM TRACK DATE 15/4/87 CODE R01BFBL0410 NGR TL 864 214	@ FEERINGBURY OLD MILL NRA CODE R01BFBL04 NGR TL 865 212
AN21	BLACKWATER	BRAXTED FGS DATE 21/4/87 NGR TL 847 161	---	@ APPLEFORD NRA CODE R01BFBL03 NGR TL 845 158
AN22	BLACKWATER	BLUEMILLS DATE 27/4/87 NGR TL 829 129	WICKHAM MILL BRIDGE DATE 6/11/86 CODE R01BFBL0160 NGR TL 824 116	@ BLUEMILLS NRA CODE R01BFBL02 NGR TL 831 132
AN23	BLACKWATER	LANGFORD DATE 5/4/87 NGR TL 836 088	B1019 RD BR LANGFORD DATE 23/4/87 CODE R01BFBL0070 NGR TL 835 910	@ LANGFORD INTAKE NRA CODE R01BFBL01 NGR TL 837 092
AN24	LITTLE OUSE	BRANDON STAUNCH DATE a. 27/9/85 b. 4/7/88 NGR TL 778 867	---	BRANDON ROAD BRIDGE NRA CODE R02BF45M04 NGR TL 784 869
AN25	LITTLE OUSE	D/S WILTON BRIDGE DATE 10/6/88 NGR TL 721 868	WILTON BRIDGE LAKENHEATH DATE 14/9/87 CODE R02BF46M01 NGR TL 724 867	WILTON BRIDGE LAKENHEATH NRA CODE R02BF46M01 NGR TL 724 867
AN26	LITTLE OUSE	LITTLE OUSE VILLAGE DATE 24/5/88 NGR TL 626 890	---	LITTLE OUSE RD BR NRA CODE R02BF46M07 NGR TL 622 893

APPENDIX B (cont.)

NRA NORTH WEST REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
NW1		REJECTED		
NW2	DOUGLAS	ABOVE SCHOLES WEIR DATE 6/7/82 NGR -	---	ABOVE SCHOLES WEIR NRA CODE 017C80576C NGR SD 586 054
NW3	DOUGLAS	WANES BLADES BRIDGE DATE 15/8/85 NGR -	---	WANES BLADES BRIDGE NRA CODE 017C80597C NGR SD 476 125
NW4	TAME	@ PORTWOOD DATE 26/7/84 NGR -	-	@ PORTWOOD NRA CODE 0169800470 NGR SJ 900 913
NW5	BOLLIN	@ MILL LANE BRIDGE, MOTTRAM DATE 16/5/84 NGR -	-	@ MILL LANE BRIDGE, MOTTRAM NRA CODE 0169802370 NGR SJ 881 803

NRA SEVERN TRENT REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
ST1	SENCE	RATCLIFFE CULEY BRIDGE DATE 3/12/80 NGR SP 317 995	RATCLIFFE CULEY DATE 20/10/80 NRA CODE 533 NGR SP 322 996	RATCLIFFE CULEY NRA CODE 358 NGR SP 322 996
ST2	MEASE	CROXALL DATE 28/10/85 NGR SK 193 138	CROXHALL DATE 23/9/85 NRA CODE 549 NGR SK 192 138	CROXALL NRA CODE 367 NGR SK 192 139
ST3		REJECTED		
ST4	PERRY	REDNAL MILL DATE a. 1/10/86 b. 27/1/87 NGR SJ 373 294	REDNAL DATE a. 24/7/86 b. 27/1/87 NRA CODE 48 NGR SJ 374 294	REDNAL NRA CODE 41 NGR SJ 373 294
ST5	PERRY	WYKEY DATE a. 7/10/86 b. 14/9/87 NGR SJ 396 242	---	WYKEY NRA CODE 42 NGR SJ 396 245
ST6	RODEN	WITHINGTON DATE 6/8/86 NGR SJ 593 143	RODDINGTON DATE 19/6/86 NRA CODE 72 NGR SJ 589 143	RODDINGTON NRA CODE 77 NGR SJ 590 143

APPENDIX B (cont.)

ST7	LEIGH BROOK	BROCKAMIN HOUSE DATE 30/10/86 NGR SO 781 534	LEIGH DATE 19/5/86 NRA CODE 138 NGR SO 781 535	LEIGH ROAD BRIDGE NRA CODE 128 NGR SO 781 534
ST8	FINHAM BROOK	FINHAM DATE 29/9/86 NGR SP 330 739	FINHAM DATE 23/9/86 NRA CODE 170 NGR SP 331 741	FINHAM BRIDGE NRA CODE 165 NGR SP 331 740
ST9	MAUN	WHITEWATER BRIDGE DATE a. 2/8/83 b. 14/6/84 c. 18/9/85 d. 10/7/86 NGR SK 663 703	WHITEWATER DATE a. 14/7/83 b. 7/6/84 c. 15/7/85 d. 15/7/86 NRA CODE 787 NGR SK 663 702	WHITEWATER NRA CODE 555 NGR SK 663 702
ST10	SOW	MILFORD DATE 9/9/82 NGR SJ 975 215	MILFORD DATE 6/9/82 NRA CODE 422 NGR SJ 975 215	MILFORD NRA CODE 268 NGR SJ 975 215
ST11	LEEN	PAPPLEWICK LIDO U/S DATE a. 17/10/84 b. 11/9/85 NGR SK 548 505	---	PAPPLEWICK MOOR NRA CODE 518 NGR SK 548 502
ST12	TEAN	CHECKLEY DATE 8/10/86 NGR SK 033 375	---	CHECKLEY NRA CODE 405 NGR SK 034 374
ST13	DERWENT	U/S HATHERSAGE WTW DATE a. 16/9/82 b. 13/9/83 NGR SK 234 806	HATHERSAGE DATE a. 23/8/82 b. 2/8/83 NRA CODE 580 NGR SK 233 806	HATHERSAGE NRA CODE 418 NGR SK 233 806
ST14	SOAR	CROFT DATE a. 13/11/80 b. 12/10/82 c. 5/7/83 d. 4/4/84 NGR SP 508 956	CROFT DATE a. -- b. -- c. 19/6/83 d. 9/11/83 NRA CODE 611 NGR SP 512 959	CROFT NRA CODE 455 NGR SP 511 959
ST15	SOAR	CROFT (REGRADED) DATE 4/4/84 NGR SP 507 953	CROFT DATE 9/11/83 NRA CODE 611 NGR SP 512 959	CROFT NRA CODE 455 NGR SP 511 959
ST16	WREAKE	QUENIBOROUGH BROOK D/S DATE 31/10/85 NGR SK 627 131	RATCLIFFE DATE 10/10/85 NRA CODE 669 NGR SK 631 142	LEWIN BRIDGE NRA CODE 489 NGR SK 622 129

APPENDIX B (cont.)

ST17	WREAKE	FRISBY DATE a. 24/4/80 b. 7/6/84 c. 24/10/85 NGR SK 684 174	FRISBY DATE a. -- b. -- c. 10/10/85 NRA CODE 668 NGR SK 686 178	FRISBY NRA CODE 487 NGR SK 686 178
ST18	IDLE	MISTERTON DATE 28/9/83 NGR SK 765 962	---	MISTERTON NRA CODE 562 NGR 766 962
ST19	MEDEN	THORESBY ESTATE DATE a. 17/7/80 b. 6/10/83 NGR SK 657 718	THORESBY DATE a. 4/3/80 b. 13/9/83 NRA CODE 797 NGR SK 648 711	THORESBY NRA CODE 569 NGR SK 648 711
ST20	MEDEN	WARSOP MILL DATE 9/12/82 NGR SK 568 687	WARSOP MILL DATE 17/9/82 NRA CODE 796 NGR SK 568 686	WARSOP MILL NRA CODE 567 NGR SK 568 686
ST21	MEASE	HARLASTON DATE 6/11/85 NGR SK 214 115	---	HARLASTON NRA CODE 366 NGR SK 215 112
ST22	REJECTED			
ST23	PERRY	D/S RUYTON WRW DATE a. 1/10/86 b. 14/9/87 NGR SJ 404 218	---	PLATT BRIDGE NRA CODE 43 NGR SJ 403 223
ST24	PERRY	MILFORD DATE a. 1/10/86 b. 15/9/87 NGR 421 210	---	MILFORD NRA CODE 44 NGR SJ 422 211
ST25	PERRY	MYTTON MILL DATE a. 30/9/86 b. 1/9/87 NGR SJ 440 173	MYTTON DATE a. 30/9/86 b. 11/8/87 NRA CODE 49 NGR SJ 439 171	MYTTON NRA CODE 45 NGR SJ 439 170
ST26	PERRY	FITZ DATE a. 30/9/86 b. 15/9/87 NGR SJ 443 180	MYTTON DATE a. 30/9/86 b. 11/8/87 NRA CODE 49 NGR SJ 439 171	MYTTON NRA CODE 45 NGR SJ 439 170

APPENDIX B (cont.)

NRA SOUTH WEST REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
SW1	OKEMENT	IDDESLEIGH DATE 29/6/83 NGR SS 567 057	---	IDDESLEIGH BRIDGE NRA CODE 09R29D006 NGR SS 567 058
SW2	OKEMENT	WOODHALL BRIDGE DATE 28/6/83 NGR SS 584 034	WOODHALL DATE -/-/83 NGR SS 584 038	WOODHALL BRIDGE NRA CODE 09R29D005 NGR SS 584 034
SW3	TAVY	HILLBRIDGE DATE 27/7/83 NGR SX 535 806	---	HILL BRIDGE NRA CODE R12C001 NGR SX 532 804
SW4	TAVY	HARFORD BRIDGE DATE 2/8/83 NGR SX 505 769	---	@ HARFORD BRIDGE NRA CODE R12C002 NGR SX 506 768
SW5	TAVY	LANGHAM WOOD DATE 8/9/83 NGR SX 466 712	---	@ WASHFORD NRA CODE R12C005 NGR SX 470 711
SW6	WALLABROOK (TAVY)	WALLABROOK DATE 4/8/83 NGR SX 502 765	---	PRIOR TO CONFLUENCE WITH TAVY NRA CODE R12C011 NGR SX 492 755
SW7	LUMBURN (TAVY)	SHILLAMILL DATE 29/7/83 NGR SX 467 719	---	@ SHILLAMILL P.T.C. WITH TAVY NRA CODE R12C010 NGR SX 467 719
SW8	WALKHAM (TAVY)	WARD BRIDGE DATE 8/7/83 NGR SX 5435 7235	---	WARD BRIDGE NRA CODE R12D002 NGR SX 542 721
SW9	MILTON BROOK (TAVY)	MILTON COMBE STREAM DATE 13/9/83 NGR SX 481 647	---	D/S MILTON COMBE NRA CODE R12B001 NGR SX 482 648

NRA SOUTHERN REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
SO1	OUSE	ARDINGLY GAUGING STN DATE 10/7/78 NGR TQ 332 283	---	BY MSW COY @ ARDINGLY NRA CODE 071002830400200 NGR TQ 332 283
SO2	OUSE	THE SLOOP DATE 1/9/78 NGR TQ 386 245	---	FRESHFIELD BRIDGE NRA CODE 071002810400130 NGR TQ 385 245
SO3	OUSE	WOODGATE DAIRIES DATE 7/10/85 NGR TQ 407 233	---	SHEFFIELD PARK STN NRA CODE 071002810400150 NGR NGR TQ 406 236

APPENDIX B (cont.)

SO4	OUSE	REJECTED		
SO5	UCK	DATE 20/3/84 NGR TQ 460 203	---	OWLSBURY FARM NRA CODE 071002910400090 NGR TQ 459 202
SO6	CUCKMERE	COWBEECH PUMPING STN DATE 12/3/84 NGR TQ 610 150	SHEEPWASH BRIDGE DATE 18/1/84 NGR TQ 611 152	SHEEPWASH BRIDGE NRA CODE 071001550400020 NGR TQ 611 150
SO7	CUCKMERE	WALDRON GHYLL DATE 13/3/84 NGR TQ 595 161	---	WALDRON GHYLL NRA CODE 071001550400050 NGR TQ 595 161
SO8	BREDE	D/S BREDE BRIDGE DATE 17/4/85 NGR TQ 828 175	BREDE BRIDGE DATE 25/3/85 NGR TQ 826 175	BREDE BRIDGE NRA CODE 070006320406540 NGR TQ 827 176
SO9	ARUN	HORSHAM BY-PASS DATE 27/2/84 NGR TQ 150 299	---	@ F/B D/S OF HORSHAM BY-PASS (A24) NRA CODE 071004270400160 NGR TQ 151 300
SO10	ARUN	NEW BRIDGE DATE 12/10/83 NGR TQ 068 259	---	NEW BRIDGE NRA CODE 071004230400070 NGR TQ 069 260
SO11	ADUR (EAST)	U/S BURGESS HILL STW DATE a. 18/4/78 b. 18/7/83 NGR TQ 311 207	---	a. U/S BURGESS HILL STW NRA CODE 071003230400020 NGR TQ 311 207 b. F/B ABOVE BURGESS HILL STW NRA CODE 071003230400040 NGR TQ 315 211
SO12	ADUR (EAST)	FAIRPLACE BRIDGE DATE 19/4/78 NGR TQ 303 204	---	FAIRPLACE BRIDGE NRA CODE 071003230400030 NGR TQ 306 204

NRA THAMES REGION

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
TH1	LODDON	Priory farm DATE 14/9/86 NGR SU 704 639	---	Kings Bridge NRA CODE LDR.0028 NGR SU 715 647
TH2	LODDON	Twyford left channel DATE 18/9/86 NGR SU 782 762	---	@ A4 roadbridge, Twyford NRA CODE LDR.0029 NGR SU 779 766
TH3	BLACKWATER	BLN8 DATE 23/9/85 NGR SU 730 650	@ gauging stn, Swallowfield DATE 27/2/85 NGR SU 731 648	@ gauging stn., Swallowfield NRA CODE LDR.0010 NGR SU 731 648

APPENDIX B (cont.)

TH4	BLACKWATER	BLP2 DATE 17/10/85 NGR ~SU 879 565	@ Frimley Bridges DATE 6/2/86 NGR SU 872 577	@ Frimley Bridges NRA CODE LDR.0009 SU 872 577
TH5	THAME	Lower Hartwell DATE a. 25/7/85 b. 6/10/87 NGR SP 770 135	Above Eythrope Lake DATE a. 7/3/85 b. 16/9/87 NGR SP 776 130	Above Eythrope Lake NRA CODE - NGR SP 776 135
TH6	RODING	High Ongar DATE SPRING 77 NGR TL 561 042	@ High Ongar Bridge DATE 10/1/77 NGR --	d/s gauging weir, High Ongar NRA CODE -- NGR TL 561 041
TH7	RODING	Theydon Bois DATE SPRING 77 NGR TQ 478 977	---	Abridge NRA CODE -- NGR TQ 467 971
TH8	RODING	d/s Chigwell DATE SPRING 77 NGR ~TQ 417 910	@ Woodford DATE 27/6/77 NGR --	Woodford Bridge NRA CODE -- NGR TQ 418 916

TWEED RIVER PURIFICATION BOARD

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
TRPB1	BILLIE BURN (TWEED)	BILLIE BURN DATE a. 12/7/88 b. 21/9/88 NGR NT 851 566	BILLIE BURN FOOT DATE a. 27/5/88 b. 19/9/88 NGR NT 851 566	BILLIE BURN AT FOOT NRA CODE -- NGR NT 851 565
TRPB2	LEET WATER (TWEED)	LEET WATER DATE 13/7/88 NGR NT 814 413	CHARTERPATH BRIDGE DATE 9/12/87 NGR NT 814 413	@ CHARTERPATH BRIDGE NRA CODE -- NGR NT 814 413
TRPB3	EDEN WATER (TWEED)	EDEN WATER DATE 13/7/88 NGR NT 739 372	---	@ PIPERS GRAVE NRA CODE -- NGR NT 728 373
TRPB4	JED WATER (TWEED)	MOUNTHOOLY DATE 29/8/88 NGR NT 662 238	JED WATER FOOT DATE 8/3/88 NGR NT 661 242	JED WATER FOOT NRA CODE -- NGR NT 661 240
TRPB5	JED WATER (TWEED)	GLEBE DATE 29/8/88 NGR NT 650 203	ABBEY BRIDGE DATE 9/3/88 NGR NT 650 203	@ ABBEY BRIDGE NRA CODE -- NGR NT 651 203
TRPB6	HEADSHAW BURN (TWEED)	BELOW BRIDGE DATE 11/8/88 NGR NT 493 547	BELOW A68 DATE 28/1/88 NGR NT 494 545	@ A68 NRA CODE -- NGR NT 494 545
TRPB7	GALA WATER (TWEED)	BOWMOUNT DATE 12/8/88 NGR NT 455 401	---	@ BOWLAND BRIDGE NRA CODE -- NGR NT 455 401
TRPB8	GALA WATER (TWEED)	BOWER 1 DATE 11/8/88 NGR NT 428 501	---	ABOVE FOUNTAINHALL NRA CODE -- NGR 427 498

APPENDIX B (cont.)

TRPB9	GALA WATER (TWEED)	BOWER 2 DATE 11/8/88 NGR NT 429 501	---	ABOVE FOUNTAINHALL NRA CODE -- NGR 427 498

FORTH RIVER PURIFICATION BOARD

SITE CODE	RIVER	FISHERIES SITE	INVERTEBRATE SITE	WATER QUALITY SITE
FRPB1	ALLAN	DUNBLANE DATE 16/7/86 NGR NN 781 011	---	@ DUNBLANE NRA CODE -- NGR NN 782 010
FRPB2	ALMOND	ALMOND DATE 24/9/86 NGR NT 180 755	D/S DOWIE'S MILL WEIR DATE 19/9/86 NGR NT 178 757	@ CRAMOND BRIDGE NRA CODE -- NGR --

APPENDIX C - FREQUENCY OF OCCURRENCE OF ENVIRONMENTAL VARIABLES WITHIN THE DATABASE

Notes on Appendix C:

i) Sites

Site Codes are as defined in Appendix B.

SO=Southern NRA Region, TH=Thames NRA Region, SW=South west NRA Region, AN=Anglian NRA Region, ST=Severn Trent NRA Region, NW=North West NRA Region, TRPB=Tweed River Purification Board, FRPB=Forth River Purification Board.

ii) Chemical Determinands

pH = pH
Cond = Conductivity ($\mu s\ cm^{-1}$)
Temp = Temperature (°C)
SSLDS = Suspended Solids ($mg\ l^{-1}$)
TDS = Total Dissolved Solids ($mg\ l^{-1}$)
BOD = Biological Oxygen Demand ATU ($mg\ l^{-1}$)
DO = Dissolved Oxygen ($mg\ l^{-1}$)
NH_3N = Total Ammoniacal Nitrogen ($mg\ l^{-1}$)
uNH_3N = Unionised Ammonia ($mg\ l^{-1}$)
NO_2N = Nitrite Nitrogen ($mg\ l^{-1}$)
NO_3N = Nitrate Nitrogen ($mg\ l^{-1}$)
PO_4 = Orthophosphate ($mg\ l^{-1}$)
CN = Total Cyanide ($mg\ l^{-1}$)
Thio = Thiocyanate ($mg\ l^{-1}$)
Hard = Hardness ($mg\ l^{-1}\ CaCO_3$)
Alky = Alkalinity ($mg\ l^{-1}\ CaCO_3$)
CO_2 = Free Carbon Dioxide ($mg\ l^{-1}$)
Cl_2 = Chlorine ($mg\ l^{-1}$)
Al t = Total Aluminium ($mg\ l^{-1}$)
Al s = Soluble Aluminium ($mg\ l^{-1}$)
Fe t = Total Iron ($mg\ l^{-1}$)
Fe s = Soluble Iron ($mg\ l^{-1}$)
Cr t = Total Chromium ($mg\ l^{-1}$)
Cr s = Soluble Chromium ($mg\ l^{-1}$)
Zn t = Total Zinc ($mg\ l^{-1}$)
Zn s = Soluble Zinc ($mg\ l^{-1}$)
Ni t = Total Nickel ($mg\ l^{-1}$)
Ni s = Soluble Nickel ($mg\ l^{-1}$)
Cu t = Total Copper ($mg\ l^{-1}$)
Cu s = Soluble Copper ($mg\ l^{-1}$)
Cd t = Total Cadmium ($mg\ l^{-1}$)
Cd s = Soluble Cadmium ($mg\ l^{-1}$)
Pb t = Total Lead ($mg\ l^{-1}$)
Pb s = Soluble Lead ($mg\ l^{-1}$)
Hg t = Total Mercury ($mg\ l^{-1}$)
As t = Total Arsenic ($mg\ l^{-1}$)
As s = Soluble Arsenic ($mg\ l^{-1}$)
Phen = Total Phenols ($mg\ l^{-1}$)
Sdet = Synthetic Detergents ($mg\ l^{-1}$)
Hcb = Hydrocarbons ($mg\ l^{-1}$
PAH = Polycyclic Aromatic Hydrocarbons ($mg\ l^{-1}$)

Other toxicant determinands considered but for which no data was received:

Haloforms and pesticides.

Symbols used:

* indicates at least 5 analytical results available over period chosen.
+ indicates less than 5 (but more than 0) analytical results available over period chosen.

iii) Habitat and Biological Variables

Width	= Mean Width of River at Site Sampled
Depth	= Mean Depth of River at Site Sampled
Altitude	= Estimated Altitude of Site Sampled
Slope	= Estimated Gradient of River at Site Sampled
BMWP	= Biological Monitoring Working Party Score
ASPT	= Average Score Per Taxon

Site Code	WATER QUALITY PARAMETERS												
	pH	Cond uScm-1	Temp °C	SSLDS	TDS	BOD	DO	NH_3N (mg l^{-1})	uNH_3N	NO_2N	NO3N	PO_4	CN
S01	*	*	*	*		*	*	*	*	*	*	*	
S02	*	*	*	*		*	*	*		*	*	*	
S03	*	*	*			*	*	*		*	*	*	
S05	*	*	*			*	*	*		*	*	*	
S06	*	*	*			*	*	*		*	*	*	
S07	*	*	*			*	*	*		*	*	*	
S08	*	*	*			*	*	*		*	*	*	
S09	*	*	*			*	*	*		*	*	*	
S010	*	+	*			*	*	*		*	*	*	
S011a	*	*	*				*	*		*	*	*	
S011b	*	+	*	+		*	*	*		*	*	+	
S012	*	*	*				*	*		*	*	*	
TH1	*		*			*	*	*	*			*	
TH2	*		*			*	*	*	*			*	
TH3	*		*	*		*	*	*	*			*	
TH4	*		*	*		*	*	*	*			*	
TH5a	*		*	*		*	*	*	*				
TH5b	*		*			*	*	*	*				
TH6	*	+	*			*	*	*			*	*	
TH7	*	+	*	*		*	*	*		*	*	*	
TH8	*	+	*	*		*	*	*	*	*	*	*	
SW1	*	*	*	*		*	*	*	*	*	*	*	
SW2	*	*	*	*		*	*	*	*	*	*	*	
SW3	*	*	*	*		*	*	*	*	*	*	*	
SW4	*	*	*	*		*	*	*	*	*	*	*	
SW5	*	*	*	*		*	*	*	*	*	*	*	
SW6	*	*	*	*		*	*	*	*	*	*	*	
SW7	*	*	*	*		*	*	*	*	*	*	*	
SW8	*	*	*	*		*	*	*	*	*	*	*	
SW9	*	*	*	*		*	*	*	*	*	*	*	
AN1	*		*	*		*	*	*		*	*		
AN2	*		*	*		*	*	*		*	*		
AN3	*		*	*		*	*	*		*	*		
AN5	*		*	*		*	*	*		*	*	+	
AN6	*		*	*		*	*	*		+	+		
AN7	*		*	*		*	*	*		*	*		
AN8	*		*	*		*	*	*		*	*		
AN9	*		*	*		*	*	*		*	*		
AN11	*		*	*		*	*	*		*	*		
AN12	*		*	*		*	*	*		*	*		
AN13	*		*	*		*	*	*		*	*		
AN14	*		*	*		*	*	*		*	*		
AN15	*		*	*		*	*	*		*	*		
AN16	*	*	*			*	*	*				*	
AN17	*	*	*			*	*	*				*	

Site Code	WATER QUALITY PARAMETERS												
	pH	Cond uScm-1	Temp °C	SSLDS	TDS	BOD	DO	NH_3N ($mg\ l^{-1}$)	uNH_3N	NO_2N	NO3N	PO_4	CN
AN18	*	*	*	*		*	*	*				*	
AN19	*	*	*			*	*	*				*	
AN20	*	*	*			*	*	*				*	
AN21	*	*	*			*	*	*				*	
AN22	*	*	*			*	*	*				*	
AN23	*	*	*			*	*	*		*		*	
AN24a	*		*	*		*	*	*		*			
AN24b	*		*	*		*	*	*		*			
AN25	*		*			*	*	*					
AN26	*		*	*		*	*	*		*			
ST1	*	*	*	*		*	*	*				*	
ST2	*	*	*	*		*	*	*	*			*	
ST4a	*	*	*	*		*	*	*	*				
ST4b	*	*	*	*		*	*	*	*				
ST5a	*	*	*	*		*	*	*	*				
ST5b	*	*	*	*		*	*	*	*				
ST6	*	*	*	*		*	*	*	*			*	
ST7	*	*	*	*		*	*	*	*				
ST8	*	*	*	*		*	*	*	*				
ST9a	*	*	*	*		*	*	*	*			*	
ST9b	*	*	*	*		*	*	*	*			*	
ST9c	*	*	*	*		*	*	*	*			*	
ST9d	*	*	*	*		*	*	*	*			*	
ST10	*	*	*	*		*	*	*	*			*	
ST11a	*	*	*	*		*	*	*	*				
ST11b	*	*	*	*		*	*	*	*				
ST12	*	*	*	*		*	*	*	*				
ST13a	*	*	*	*		*	*	*	*				
ST13b	*	*	*	*		*	*	*	*				
ST14a	*	*	*	*		*	*	*	*				
ST14b	*	*	*	*		*	*	*	*				
ST14c	*	*	*	*		*	*	*	*				
ST14d	*	*	*	*		*	*	*	*				
ST15	*	*	*	*		*	*	*	*				
ST16	*	*	*	*		*	*	*	*				
ST17a	*	*	*	*		*	*	*	*				
ST17b	*	*	*	*		*	*	*	*				
ST17c	*	*	*	*		*	*	*	*				
ST18	*	*	*	*		*	*	*	*				
ST19a	*	*	*	*		*	*	*	*			*	
ST19b	*	*	*	*		*	*	*	*				
ST20	*	*	*	*		*	*	*	*				
ST21	*	*	*	*		*	*	*	*				
ST23a	*	*	*	*		*	*	*	*				
ST23b	*	*	*	*		*	*	*	*				

Site Code	WATER QUALITY PARAMETERS												
	pH	Cond uScm-1	Temp °C	SSLDS	TDS	BOD	DO	NH_3N (mg l^{-1})	uNH_3N	NO_2N	NO3N	PO_4	CN
ST24a	*	*	*	*		*	*	*	*				
ST24b	*	*	*	*		*	*	*	*				
ST25a	*	*	*	*		*	*	*	*			*	
ST25b	*	*	*	*		*	*	*	*			*	
ST26a	*	*	*	*		*	*	*	*			*	
ST26b	*	*	*	*		*	*	*	*			*	
NW2	*		*				*	*		*			
NW3	*		*				*	*		*			
NW4	*		*				*	*		*			
NW5	*		*				*	*		*			
TRPB1a	*	*	*	*		*	*	*			*	*	
TRPB1b	*	*	*	*		*	*	*			*	*	
TRPB2	*	*	*	*		*	*	*			*	*	
TRPB3	*	*	*	*		*	*	*			*	*	
TRPB4	*	*	*	*		*	*	*			*	*	
TRPB5	*	*	*	*		*	*	*			*	*	
TRPB6	*	*	*	*		*	*	*			*	*	
TRPB7	*	*	*	*		*	*	*			*	*	
TRPB8	*	*	*	*		*	*	*			*	*	
TRPB9	*	*	*	*		*	*	*			*	*	
FRPB1	*	*	*	*		*	*	*				*	
FRPB2	*	*	*	*		*	*	*		*		*	

Site Code	WATER QUALITY PARAMETERS (mg l^{-1})													
	Thio	Hard	Alky	CO_2	Cl_2	Al t	Al s	Fe t	Fe s	Cr t	Cr s	Zn t	Zn s	Ni t
S01		*	*											
S02			*											
S03		*	*											
S05		*	*											
S06		*	*											
S07		*	*											
S08		+	*											
S09		*	*											
S010		+	+											
S011a														
S011b		+	+											
S012														
TH1														
TH2										*		*		*
TH3										*		*		*
TH4														
TH5a														
TH5b														
TH6														
TH7			*											
TH8														
SW1		*	*					*		*		*		*
SW2		*	*					*		*		*		*
SW3		*	*	*				*		*		*		*
SW4		*	*	*				*		*		*		*
SW5		*	*	*				*		*		*		*
SW6		*	*	*				*		*		*		*
SW7		*	*	*				*		*		*		*
SW8		*	*	*				*		*		*		*
SW9		*	*	*				*		*		*		*
AN1			*											
AN2			*											
AN3			*											
AN5			*											
AN6			+											
AN7			*											
AN8			*											
AN9			*											
AN11			*											
AN12			*											
AN13			*											
AN14			*											
AN15			*											
AN16														
AN17														

Site Code	WATER QUALITY PARAMETERS													
	Thio	Hard	Alky	CO_2	Cl_2	Al t	Al s (MG L^{-1})	Fe t	Fe s	Cr t	Cr s	Zn t	Zn s	Ni t
AN18														
AN19														
AN20														
AN21														
AN22														
AN23		*	*					*						
AN24a			*											
AN24b			*											
AN25														
AN26			*											
ST1		*	*							*	*	*	*	*
ST2		*	*							*	*	*	*	*
ST4a		*	*											
ST4b		*	*											
ST5a		*	*											
ST5b		*	*											
ST6		*	*											
ST7		+	*							+	+	+	+	+
ST8		*												
ST9a										*	*	*	*	*
ST9b		*	*							*	*	*	*	*
ST9c		*	*							*	*	*	*	*
ST9d		*	*							*	*	*	*	*
ST10		*	*							*	*	*	*	*
ST11a		*												
ST11b										*	*	*	*	*
ST12														
ST13a														
ST13b														
ST14a		*	*											
ST14b														
ST14c														
ST14d														
ST15														
ST16		*	*							*		*		*
ST17a		*	*											
ST17b														
ST17c														
ST18								+	+	*	*	*	*	*
ST19a		*	*							*	*	*	*	*
ST19b														
ST20														
ST21														
ST23a		*	*											
ST23b		*	*											

Site Code	WATER QUALITY PARAMETERS (mg l^{-1})													
	Thio	Hard	Alky	CO_2	Cl_2	Al t	Al s	Fe t	Fe s	Cr t	Cr s	Zn t	Zn s	Ni t
ST24a		*	*											
ST24b		*	*											
ST25a		*	*											
ST25b		*	*											
ST26a		*	*											
ST26b		*	*											
NW2		*												
NW3										*		*		*
NW4										*		*		*
NW5										*		*		*
TRPB1														
TRPB1														
TRPB2														
TRPB3														
TRPB4														
TRPB5														
TRPB6														
TRPB7														
TRPB8														
TRPB9														
FRPB1		*	*											
FRPB2		*	*											

Site Code	WATER QUALITY PARAMETERS (mg l^{-1})													
	Ni s	Cu t	Cu s	Cd t	Cd s	Pb t	Pb s	Hg t	As t	As s	Phen	Sdet	Hcb	PAH
S01														
S02														
S03														
S05														
S06														
S07														
S08														
S09														
S010														
S011a														
S011b														
S012														
TH1														
TH2														
TH3		*		*		+								
TH4		*		*										
TH5a														
TH5b														
TH6														
TH7														
TH8														
SW1		*		*		*								
SW2		*		*		*								
SW3		*		*		*								
SW4		*		*		*								
SW5		*		*		*								
SW6		*		*		*								
SW7		*		*		*								
SW8		*		*		*								
SW9		*		*		*								
AN1														
AN2														
AN3														
AN5														
AN6														
AN7														
AN8														
AN9														
AN11														
AN12														
AN13														
AN14														
AN15														
AN16														
AN17														
AN18														

Site Code	WATER QUALITY PARAMETERS													
	Ni s	Cu t	Cu s	Cd t	Cd s	Pb t (mg l^{-1})	Pb s	Hg t	As t	As s	Phen	Sdet	Hcb	PAH
AN19														
AN20														
AN21														
AN22														
AN23														
AN24a														
AN24b				*										
AN25														
AN26														
ST1	*	*	*	*	*	*	*							
ST2	*	*	*	*	*	*	*							
ST4a														
ST4b														
ST5a														
ST5b														
ST6														
ST7	+	+		+	+	+	+							
ST8														
ST9a	*	*	*	*	*	*	*							
ST9b	*	*	*	*	*	*	*							
ST9c	*	*	*	*	*	*	*							
ST9d	*	*	*	*	*	*	*							
ST10	*	*	*	*	*	*	*		*					
ST11a														
ST11b														
ST12	*	*	*	*	*	*	*							
ST13a														
ST13b														
ST14a														
ST14b														
ST14c														
ST14d														
ST15														
ST16		*		*		*								
ST17a														
ST17b														
ST17c														
ST18	*	*	*	*	+	*	*		*					
ST19a	*	*	*	*	*	*						*		
ST19b														
ST20														
ST21														
ST23a														
ST23b														
ST24a														
ST24b														

Site Code	WATER QUALITY PARAMETERS													
	Ni s	Cu t	Cu s	Cd t	Cd s	Pb t (mg l^{-1})	Pb s	Hg t	As t	As s	Phen	Sdet	Hcb	PAH
ST25a														
ST25b														
ST26a														
ST26b														
NW2														
NW3				*		*								
NW4				*		*								
NW5				*		*								
TRPB1														
TRPB1														
TRPB2														
TRPB3														
TRPB4														
TRPB5														
TRPB6														
TRPB7														
TRPB8														
TRPB9														
FRPB1														
FRPB2														

Site	HABITAT AND BIOLOGICAL VARIABLES					
	Width m	Depth cm	Altitude m	Slope %	BMWP	ASPT
S01	*		*	*		
S02			*	*		
S03	*		*	*		
S05	*		*	*		
S06	*		*	*	*	*
S07	*		*	*		
S08	*		*	*	*	
S09	*		*	*		
S010	*		*	*		
S011a	*		*	*		
S011b	*		*	*		
S012	*		*	*		
TH1	*	*	*	*		
TH2	*	*	*	*		
TH3	*	*	*	*	*	*
TH4	*	*	*	*	*	*
TH5a	*	*	*	*	*	*
TH5b	*	*	*	*	*	*
TH7	*	*	*	*		
TH8	*	*	*	*	*	*
SW1	*		*	*		
SW2	*		*	*	*	*
SW3	*		*	*		
SW4	*		*	*		
SW5	*		*	*		
SW6	*		*	*		
SW7	*		*	*		
SW8	*		*	*		
SW9	*		*	*		
AN1	*		*	*	*	*
AN2	*		*	*	*	*
AN3	*		*	*	*	*
AN5	*		*	*		
AN6	*		*	*		
AN7	*		*	*	*	*
AN8	*		*	*		
AN9	*		*	*	*	*
AN11	*		*	*	*	*
AN12	*		*	*	*	*
AN13	*		*	*	*	*
AN14	*		*	*	*	*
AN15	*		*	*	*	*
AN16			*	*		
AN17			*	*		
AN18			*	*		
AN19			*	*		

Site	HABITAT AND BIOLOGICAL VARIABLES					
	Width m	Depth cm	Altitude m	Slope %	BMWP	ASPT
AN20			*	*	*	*
AN21			*	*		
AN22	*		*	*	*	*
AN23	*		*	*	*	*
AN24a	*	*	*	*		
AN24b	*	*	*	*		
AN25	*		*	*	*	*
AN26	*		*	*		
ST1	*	*	*	*	*	
ST2	*	*	*	*	*	
ST4a	*	*	*	*	*	
ST4b	*	*	*	*	*	
ST5a	*	*	*	*		
ST5b	*	*	*	*		
ST6	*	*	*	*	*	
ST7	*	*	*	*	*	
ST8	*	*	*	*	*	
ST9a	*	*	*	*	*	
ST9b	*	*	*	*	*	
ST9c	*	*	*	*	*	
ST9d	*	*	*	*	*	
ST10	*	*	*	*	*	
ST11a	*	*	*	*		
ST11b	*	*	*	*		
ST12	*	*	*	*		
ST13a	*	*	*	*	*	*
ST13b	*	*	*	*	*	*
ST14a	*	*	*	*		
ST14b	*	*	*	*		
ST14c	*	*	*	*	*	
ST14d	*	*	*	*	*	
ST15	*		*	*	*	
ST16	*	*	*	*	*	
ST17a	*	*	*	*		
ST17b	*	*	*	*		
ST17c	*	*	*	*	*	
ST18	*	*	*	*		
ST19a	*	*	*	*	*	
ST19b	*	*	*	*	*	
ST20	*	*	*	*	*	
ST21	*	*	*	*		
ST23a	*	*	*	*		
ST23b	*	*	*	*		
ST24a	*	*	*	*		
ST24b	*	*	*	*		
ST25a	*	*	*	*	*	

Site	HABITAT AND BIOLOGICAL VARIABLES					
	Width m	Depth cm	Altitude m	Slope %	BMWP	ASPT
ST25b	*	*	*	*	*	
ST26a	*	*	*	*	*	
ST26b	*	*	*	*	*	
NW2	*		*	*		
NW3			*	*		
NW4	*		*	*		
NW5	*		*	*		
TRPB1a	*	*	*	*	*	*
TRPB1b	*	*	*	*	*	*
TRPB2	*	*	*	*	*	*
TRPB3	*	*	*	*		
TRPB4	*	*	*	*	*	*
TRPB5	*	*	*	*	*	*
TRPB6	*		*	*	*	*
TRPB7	*	*	*	*		
TRPB8	*	*	*	*		
TRPB9	*	*	*	*		
FRPB1	*	*	*	*		
FRPB2	*	*	*	*	*	

APPENDIX D - DETAILS OF THE 'TOXIC' PROGRAM FOR THE SUMMATION OF TOXIC EFFECTS

Toxicities are calculated and summed by the program TOXIC as below.

D.1 Toxicity to rainbow trout

Ammonia

48h LC50 = fac1 * fac2 * fac3

fac1 = factor based on alkalinity and pH, calculated by two-way interpolation from Table D1

Table D1 - Factor for ammonia toxicity from Alk. (mg l^{-1} $CaCO_3$- across) and pH (down)

0	0	25	50	100	200	250	400	1000
0	370	370	345	320	319	318	317	317
6.5	370	370	345	320	319	318	317	317
6.75	255	255	200	180	179	178	177	177
7.0	175	175	133	112	105	100	95	95
7.25	133	133	92	75	60	57.5	55	55
7.5	100	100	70	53	43	39	36	36
7.75	85	85	55	39	29	25	23	23
8.0	85	85	44	29	20.8	19	16	16
8.25	85	85	44	24	16	15	11	11
8.5	85	85	44	24	12.5	10.9	9	9
8.75	85	85	44	24	12.5	10.9	8	8
15	85	85	44	24	12.5	10.9	8	8

experimental range for pH is 6.5 to 8.75
table extended to use end point if out of range

experimental range for alkalinity 25 to 400
table extended to use end point if out of range

fac2 = factor based on DO (% satn.) and free CO_2, calculated by two-way interpolation from Table D2

Table D2 - Factor for ammonia toxicity from free CO_2 ($mg\ l^{-1}$ across) and DO % (down)

0	0	2	5	10	20	100
0	0.16	0.16	0.33	0.42	0.49	0.49
20	0.16	0.16	0.33	0.42	0.49	0.49
30	0.26	0.26	0.4	0.51	0.56	0.56
40	0.38	0.38	0.49	0.58	0.63	0.63
50	0.49	0.49	0.58	0.65	0.7	0.7
60	0.59	0.59	0.66	0.71	0.77	0.77
70	0.69	0.69	0.75	0.79	0.82	0.82
80	0.8	0.8	0.82	0.86	0.9	0.9
90	0.9	0.9	0.91	0.92	0.96	0.96
100	1	1	1	1	1	1
200	1	1	1	1	1	1

experimental range for DO is 40 to 100%
results extrapolated to 20%
table extended to use end point if out of range

experimental range for free CO_2 is 2 to 20 $mg\ l^{-1}$
table extended to use end point if out of range

free CO_2 (used for factor 2 above) is obtained as follows (if not already given)

free CO_2 = 10 ** (log10(alkalinity) - 0.0555173 - pH + A -B/2)

where A and B in the above are given by

A = 3.96 + 715/(273+t) and t= temperature in °C
B = square root of (2.5 * 0.00001 * total dissolved solids)

If the total dissolved solids concentration (tds-$mg\ l^{-1}$) is missing it is calculated as follows.

tds = 0.8 * conductivity in microsiemens cm^{-1}

fac3 = 5.75 - 1.586 * log (rtemp)

experimental range for temperature is 8 to 20 °C, end point is used when out of range

Low dissolved oxygen

Note that DO differs from the other toxicity measures in that a lower level of DO corresponds to a higher level of toxicity. A toxicity score is therefore given, which is equivalent to the fraction of the 48h LC50 for the other toxins.

toxicity score = k - 2 * x

where x = DO concentration in mg l^{-1}
and k = a + b * t + c * t**2 + d * t**3 + e * t**4
and the coefficients of the polynomial in t (temp in °C) are

a = 2.0220525
b = 0.04602609
c = 0.02041304
d = - 0.00185348
e = 0.00005278

experimental range for temperature is 9 to 26, relationship is extrapolated down to 0 and the value at the end point (26) is used when the temperature exceeds 26 °C

Nitrite

48H LC50 = 10 ** (1.0159 * log10(x) - 0.4527)
where x is the chloride ion concentration in mg l^{-1}

experimental range for chloride, up to 47 mg l^{-1}

NB. As the data sets do not include chloride concentration a value of 25 mg l^{-1} is assumed and a warning is given in the results.

Metals

48h LC50 = dofactor * exp (a * log(h) + b)

dofactor = 0.3073 * log(d) - 0.4152, where d = DO as % satn.

h is the hardness as mg l^{-1} of calcium carbonate

and the coefficients a and b depend on the metal as follows:

metal	**a**	**b**
cadmium	1.491	-6.7703
chromium	1.404	-6.651
copper	0.7807	-5.1861
nickel	0.5384	1.478
lead	0.3011	-1.0074
zinc	0.6053	-2.2137

experimental range for DO 30 to 100% saturation end point value used if DO out of range

experimental range for hardness 10 to 300mg l^{-1} end point value used if hardness out of range

Cyanide

The determinand given is assumed to be HCN so this is converted to free cyanide as follows.

free cyanide = HCN * 1.0384/ (1+ exp(2.3026*(pH-9.91-0.0275*temp)

where HCN is the HCN concentration and temp is the temperature in °C.

The 48h LC50 for free cyanide is given by the following.

48h LC50 = 0.0327 * log(t) - 0.00468 where t = temp in °C.

NB. It is unclear whether the result holds for the lower temperatures. The toxicity predicted by the equation increases rapidly as the temperature falls to around 2 °C.

Phenol

48h LC50 = dofactor * (2.75 * log(t) - 1.634) where t= temp in °C.

dofactor is the same as the DO factor used for the metals toxicity

experimental range for temperature is 6 to 18 °C.
end point is used if temperature is off range

In all cases except DO the toxicity is given by the ratio

observed concentration / 48h LC50

For DO an equivalent value to this ratio is determined directly by the method shown above.

D.2 Toxicity to roach

Two measures are given for roach

1. The ratio of observed concentration / 48h LC50.
2. The ratio of observed concentration / threshold LC50 (which may be interpreted as a no-effect concentration, but is in some cases very close to the 48h LC50).

As there are insufficient data for the effect of other factors to be included the ratios are obtained in most cases from experimentally obtained LC50 values. The exceptions are given below.

Dissolved oxygen and lead. The results for rainbow trout are used. The threshold LC50 is taken to be the same as the 48h LC50 in these cases.

Ammonia. For ammonia the LC50 values are given as undissociated ammonia. The equivalent amount of total ammonia (mg l^{-1}) needed for calculating the toxicity is obtained as follows.

LC50 (total ammonia, NH3 and NH4 anions) = L * 10 ** (PKA-pH)

where L = 48h LC50 value for undissociated ammonia
PKA = 10.05467 - 0.03246 * temp
temp = temperature in °C
(no restrictions placed on the above values)

The toxicity is then given by :
observed ammoniacal-N / LC50 as total ammonia above.

The 48h LC50 and threshold LC50 values towards roach are shown below.

determinand	**ammonia**	**DO**	**NO_2N**	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**	**CN**	**phenol**
48h LC50	0.2	-	90	20	50	0.2	190	-	15	0.13	10
threshold LC50	0.2	-	10.1	0.4	32.5	0.12	3.4	-	12.2	0.11	10

D3 - Determinands Used in the Evaluation of Predicted Toxicities

Determinand																	Scores
Amm	DO	Nit	TCd	TCr	TCu	TNi	TPb	TZn	SCd	SCr	SCu	SNi	SPb	SZn	CN	Phe	
1	1	1	1	1	1	1	1	1	0	0	0	0	0	0	1	1	total T1
1	1	1	0	0	0	0	0	0	1	1	1	1	1	1	1	1	total T2
1	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	total T3

1 = determinand used
0 = determinand not used

APPENDIX E - AN INVESTIGATION OF FISHERIES IN THE TRENT CATCHMENT USING THE TOXIC PROGRAM

Water quality and fisheries information was available for the River Trent catchment (ie Rivers Trent, Dove, Derwent, Penk, Mease, Soar, Erewash, Blythe, Amber, Churnet, Cole, Anker, Sow, Sence, Tame and Rea; and Ford and Fowlea Brooks) for the years 1970 to 1973. At this time the river was notably more polluted than in recent years due to the efforts of the pollution control authorities and the closure of some industries. Water quality determinands available were; total ammonia, phenol, nickel, lead, zinc, cadmium, copper, cyanide, pH, alkalinity, temperature, total dissolved solids, dissolved oxygen, hardness and chromium. Fishery information was only available as a rather subjective classification of status, there being the following categories for fisheries of salmonid (S) and freshwater fish (C) respectively: 1, good; 2, fair; 3, poor; and 4, fishless.

The TOXIC program was used to estimate the toxicity of the river waters to rainbow trout (as a percentage of the 48h LC50). It was only considered worthwhile to do this for sites having a minimum of 10 usable records. Two sets of toxicity predictions were made on the basis of the available data. In the first (and smaller set) input determinands were ammonia, dissolved oxygen, copper, chromium, nickel and zinc. This set included 29 records at 13 sites. The second set predicted toxicity on a more limited range of determinands; ammonia, dissolved oxygen, copper and zinc. This set included 70 records at 45 sites. For the first set (Table E1) predictions of total mean percentage toxicity ranged from 15% (Trent) to 113% (Tame). Nickel and chromium contributed very little to the total toxicity, whilst copper contributed the most (average 47% of total toxicity). Ammonia and zinc each contributed to 23% of total toxicity. Dissolved oxygen levels generally contributed little directly to the sum (average of 3%) but was the most variable, with values of up to 22% (but generally 0%-1%).

Table E1 - Predictions of Mean Total Percentage Toxicity based on concentrations of NH_3N, DO, Cr, Cu, Ni and Zn. Repeat values at sites are for different years

River	Site	Mean % Toxicity (ies)
Trent	1	18 15 17
Trent	2	26
Trent	3	30
Erewash	1	27
Cole	1	52
Tame	1	30 83
Tame	2	57 43
Tame	3	113 67 51 58
Tame	4	64 46 41 43
Tame	5	45 39
Tame	6	68 60
Tame	7	45 45 38
Tame	8	59 44 40

For the second set (Table E2) predictions of total mean percentage toxicity ranged from 9% (Mease) to 169% (Tame). The pattern of contributions (Cu, 57%; Zn, 22%; NH_3N, 18%; DO, 2%) was similar to that for first set.

Table E2 - Predictions of Mean Total Percentage Toxicity based on concentrations of NH_3N, DO, Cu and Zn. Repeat values at sites are for different years.

River	Site	Mean % Toxicity (ies)
Trent	1	34
Trent	2	18 21
Trent	3	17
Trent	4	18
Trent	5	16
Trent	6	14
Trent	7	14 16
Trent	8	12 14
Trent	9	13 11
Trent	10	64 71
Trent	11	10
Trent	12	38 39 39
Trent	13	34 30
Trent	14	22 17
Dove	1	11
Dove	2	11 10
Derwent	1	24
Derwent	2	19 14 11
Derwent	3	13
Penk	1	12
Penk	2	10
Mease	1	9
Soar	1	14 14
Soar	2	17 22 25
Soar	3	24
Erewash	1	23 74
Erewash	2	19
Blythe	1	11 12
Tame	1	102 10
Tame	2	36
Tame	3	35
Tame	4	113 41
Tame	5	47 46
Amber	1	13
Churnet	1	20
Cole	1	73 15
Anker	1	16
Anker	2	49 34

Table E2 Continued

River	Site	Mean % Toxicity (ies)
Anker	3	13 12
Sow	1	11 10 11
Fowlea Brook	1	83
Ford Brook	1	54 45
Rea	1	36

The relationship between fishery status and the predicted toxicity to both rainbow trout and roach was confounded to some extent by the large number and wide range of toxicity shown by fishless sites (Figures E1-4). However regression coefficients were always positive (Table E3) and almost always significant. The amount of variance accounted for (R2) was generally low (up to 0.41). Less significant correlations were obtained when metals were excluded from the toxicity calculations and when the 95%iles of predicted toxicity were used.

Table E3 - Regression Coefficients. Toxicity based on NH_3N, DO, Cu and Zn concentrations from sites with at least 10 complete records

Median Predicted 48h LC50 to Species	Fishery Type (S, C)	Slope	R-Squared (%)	Significance (s, ns)
Trout	S	7.4	6	s
Trout	C	7.8	41	s
Roach	S	10.6	10	s
Roach	C	7.6	33	s

The proportions of trout and roach toxicity corresponding to fishlessness were:

0.27 for trout toxicity and no salmonid fisheries
0.36 for trout toxicity and no freshwater fisheries
0.39 for roach toxicity and no salmonid fisheries
0.27 for roach toxicity and no freshwater fisheries

The values based on trout toxicity are reasonably consistent with the boundary value of 0.28 separating fish-containing from fishless sites in the previous study of the River Trent using the predicted toxicity to trout (Alabaster et al 1972).

This study of the Trent catchment adds some support to the use of the toxicological-based approach, an approach that requires a range of water and fisheries quality to demonstrate its application. This study would have been improved by using more quantitative fisheries information and possibly by transformations of the data, eg to account for the effect of variations in flow on water quality. Also, as the sites covered a range of habitat, appropriate variables, such as altitude and slope, should be included in a multiple regression analysis.

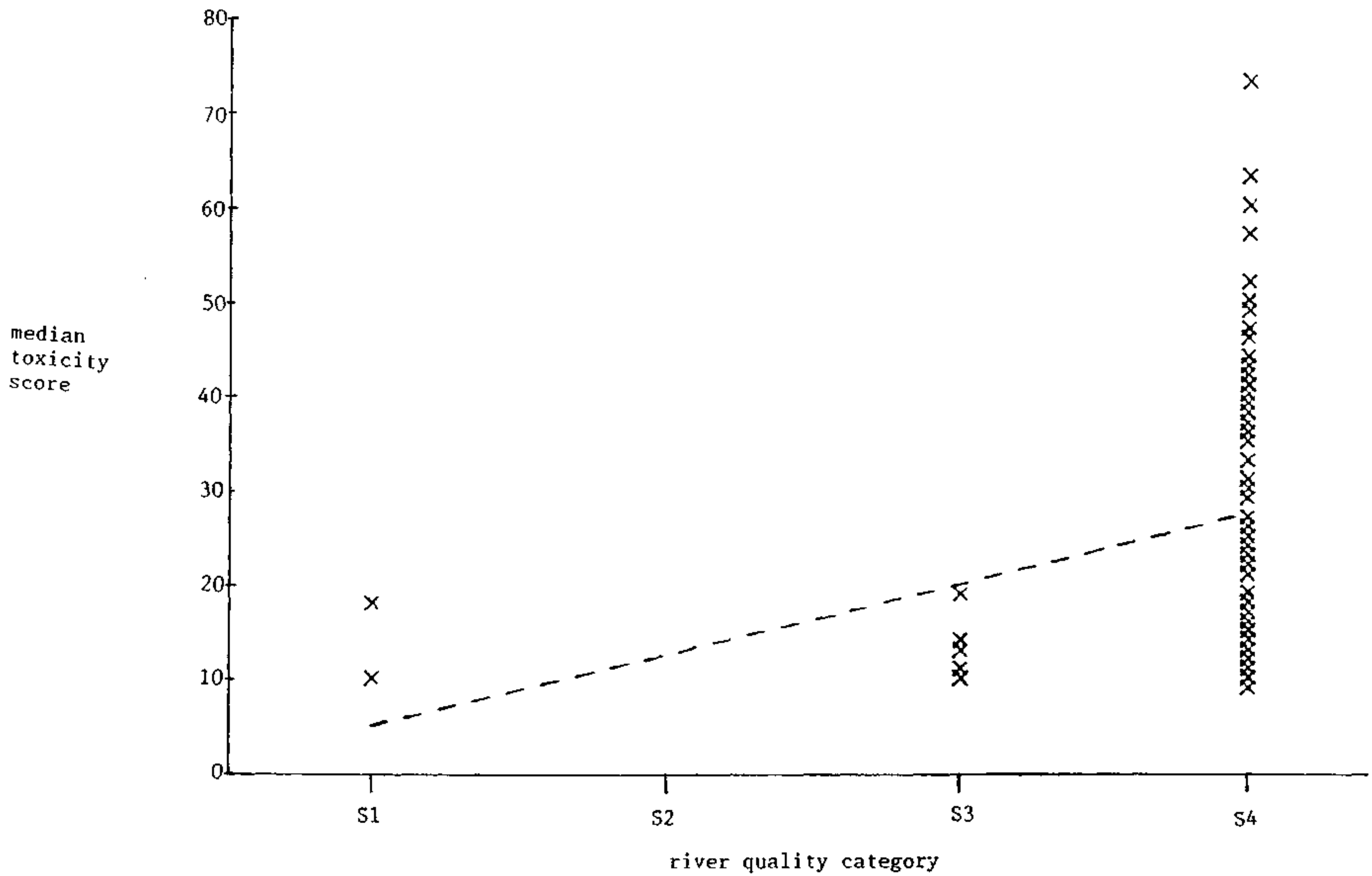


Fig E1 Predicted toxicity towards Trout (48h LC50) based on concentrations of NH_3, DO, Cu and Zn. Sites with at least 10 complete records.

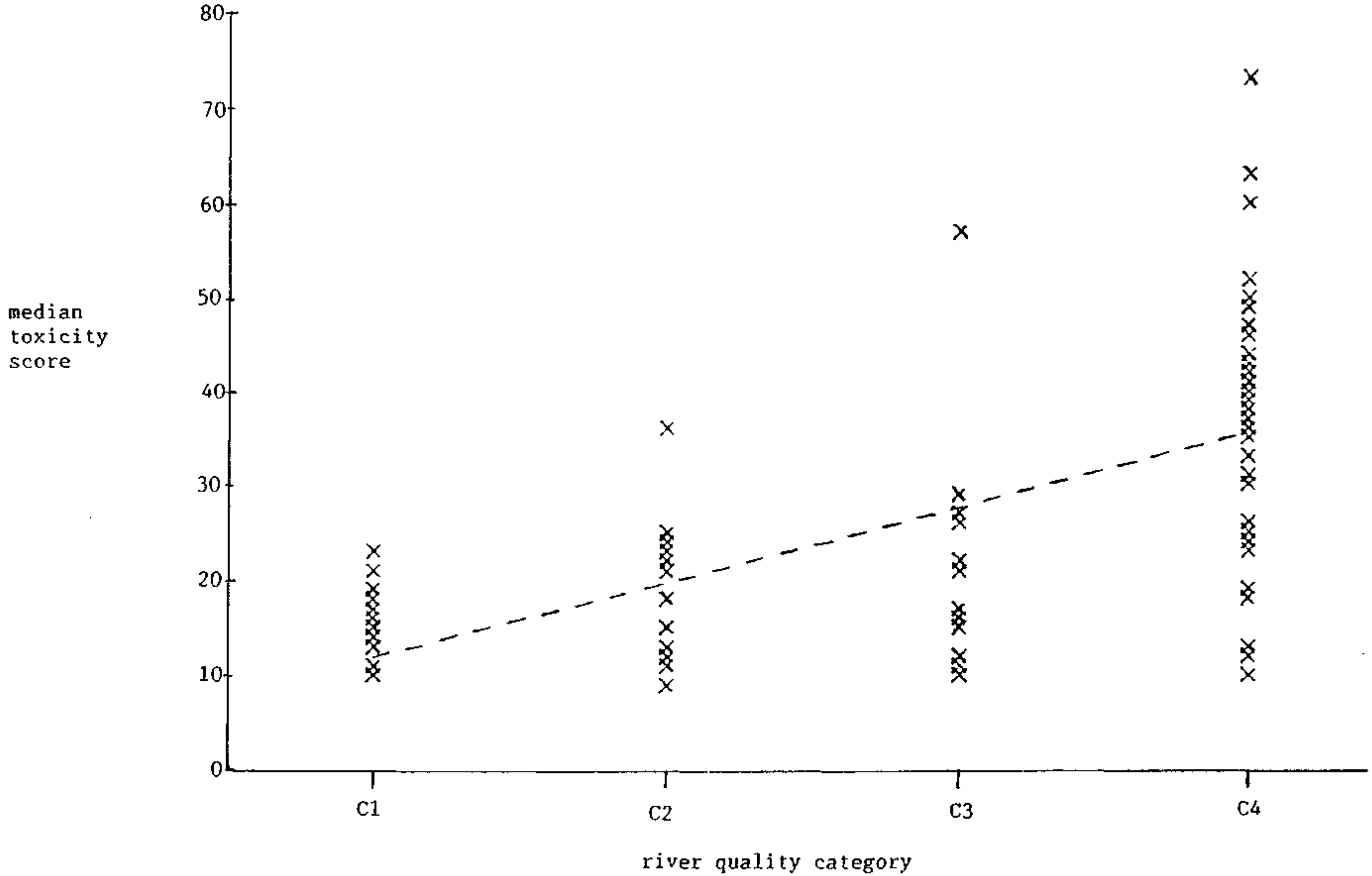


Fig E2 Predicted toxicity towards Trout (48h LC50) based on concentrations of NH3, DO, Cu and Zn. Sites with at least 10 complete records.

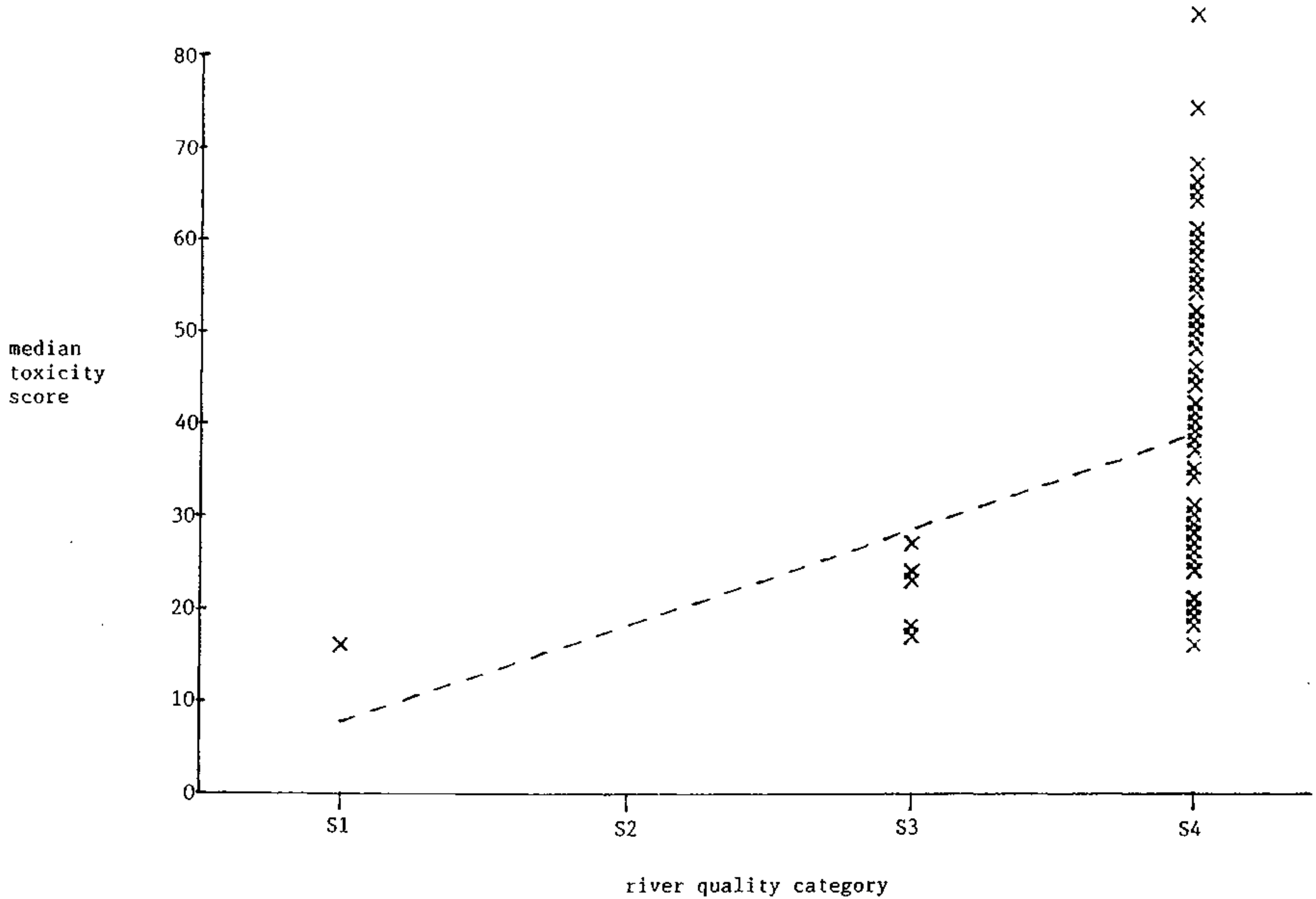


Fig E3 Predicted toxicity towards Roach (48h LC50) based on concentrations of NH_3, DO, Cu and Zn. Sites with at least 10 complete records.

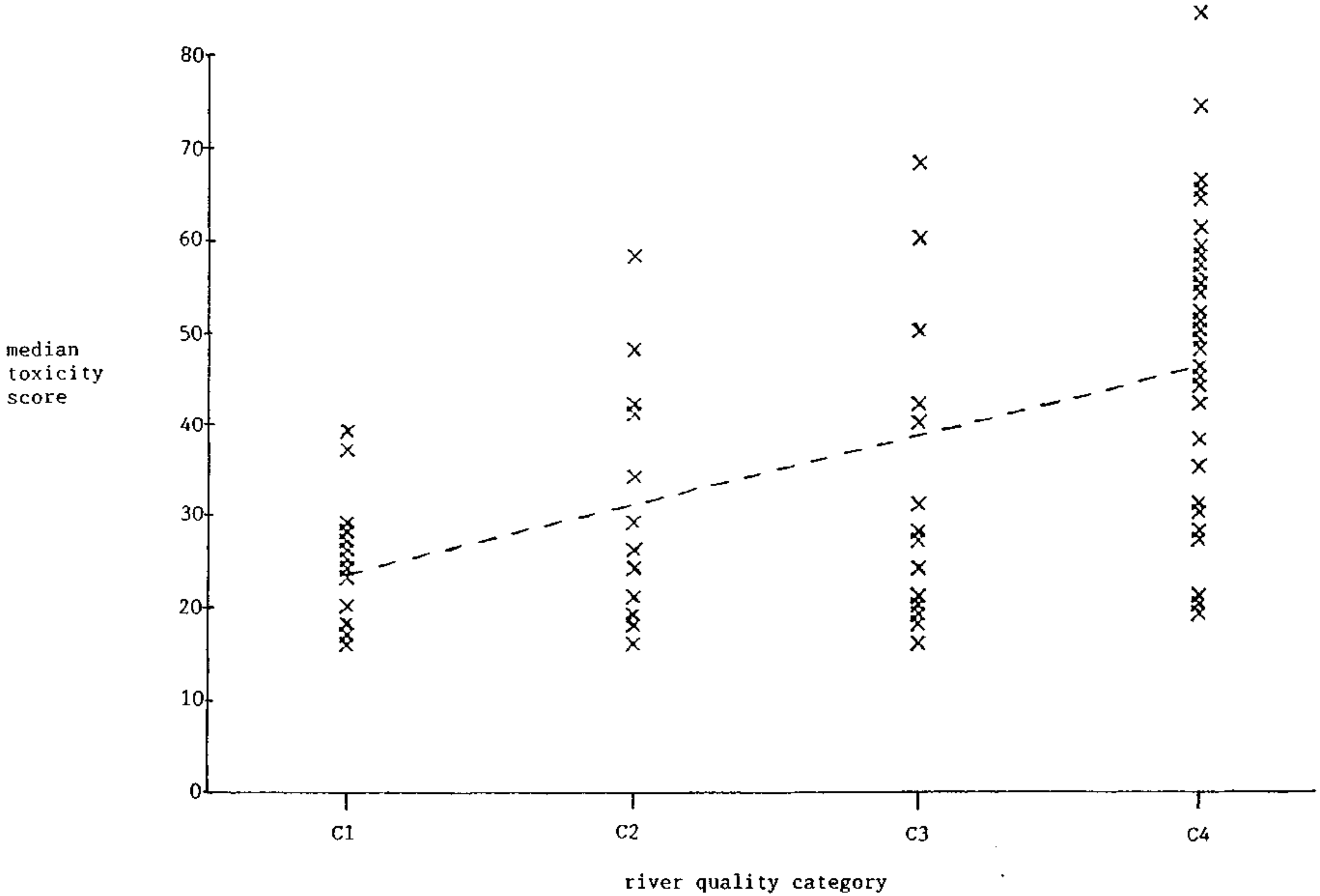


Fig E4 Predicted toxicity towards Roach (48h LC50) based on concentrations of NH_3, DO, Cu and Zn. Sites with at least 10 complete records.

APPENDIX F - BIOMASS OF SELECTED SPECIES AT EACH SITE

Site Code	Fish Biomass (g $100m^{-2}$)							
	Trout	Salmon	Pike	Perch	Chub	Dace	Roach	Gudgeon
SO1	367	0	0	119	194	148	0	5
SO2	33	0	0	1836	302	3424	8	586
SO3	52	0	0	31	449	769	63	535
SO5	437	0	0	26	3241	535	222	332
SO6	964	0	0	23	250	0	40	0
SO7	380	0	0	0	0	0	0	0
SO8	15	0	16	44	76	547	0	54
SO9	0	0	95	4	31	447	251	23
SO10	0	0	2149	45	2438	314	2899	211
SO11a	0	0	496	184	0	0	112	9
SO11b	0	0	237	2	0	0	587	44
SO12	0	0	0	0	0	0	0	0
TH1	0	0	460	0	30	70	260	10
TH2	0	0	60	0	280	30	100	30
TH3	0	0	220	0	240	100	80	10
TH4	0	0	0	0	0	10	240	0
TH5a	0	0	132	13	0	75	4185	17
TH5b	0	0	671	16	0	26	2956	2
TH6	0	0	534	1132	3	0	51	0.3
TH7	0	0	241	0	633	319	140	66
TH8	0	0	0	0	5	0	0	231
SW1	467	147	0	0	0	0	0	0
SW2	549	268	0	0	0	0	0	0
SW3	662	523	0	0	0	0	0	0
SW4	631	1195	0	0	0	0	0	0
SW5	1003	701	0	0	0	0	0	0
SW6	2184	0	0	0	0	0	0	0
SW7	1399	193	0	0	0	0	0	0
SW8	537	416	0	0	0	0	0	0
SW9	2056	100	0	0	0	0	0	0
AN1	0	0	30	0	0	0	23	14
AN2	0	0	0	0	0	0	0	0
AN3	0	0	0	10	0	0	6	6
AN5	0	0	274	6	0	0.5	466	1
AN6	0	0	117	6	4	0	1206	23
AN7	0	0	229	56	0	0	534	4
AN8	0	0	0	0	0	0	3	9
AN9	0	0	0	0	0	0	0	0
AN11	0	0	0	0	0	0	0	0
AN12	0	0	0	471	5	0	1453	100
AN13	0	0	0	8	0	0	938	264
AN14	114	0	0	0	1779	0	1715	0
AN15	274	0	0	0	1755	565	4	20
AN16	0	0	0	80	305	413	181	20
AN17	0	0	0	27	0	222	1527	67
AN18	0	0	0	0	104	162	1124	81
AN19	0	0	264	88	309	15	194	12

Site Code	Fish Biomass (g 100m⁻²) Trout	Salmon	Pike	Perch	Chub	Dace	Roach	Gudgeon
AN20	0	0	0	0	916	1716	0	3
AN21	0	0	0	0	56	24	0	0
AN22	0	0	0	0	822	512	415	131
AN23	0	0	440	15	70	0	5	0
AN24a	0	0	40	0.05	0	10	0	0
AN24b	0	0	58	2	0	13	197	0
AN25	0	0	49	13	0	0	443	0
AN26	0	0	482	6	0	11	998	2
ST1	0	0	0	66	1365	499	285	*
ST2	0	0	1294	250	4578	414	2486	*
ST4a	0	0	314	44	543	695	1	279
ST4b	0	0	55	0	25	71	0.05	49
ST5a	0	0	121	0	31	0	0	6
ST5b	0	0	145	0	166	0	0	1
ST6	0	0	575	0	5877	93	22	4
ST7	328	0	0	0	833	691	0	6
ST8	275	0	0	0	0	0	0	0
ST9a	0	0	3	35	0	79	14	660
ST9b	0	0	0	23	20	338	13	333
ST9c	0	0	0	14	125	349	9	611
ST9d	0	0	0	0	338	120	19	346
ST10	0	0	0	9	19	28	34	14
ST11a	1320	0	720	192	456	0	0	115
ST11b	489	0	598	105	569	389	0	45
ST12	0	0	0	0	82	132	0	0
ST13a	998	0	0	0	0	0	0	0
ST13b	1274	0	0	0	0	0	0	0
ST14a	0	0	156	0	428	258	820	*
ST14b	0	0	0	7	535	152	1660	*
ST14c	0	0	33	62	236	142	1505	*
ST14d	0	0	335	87	1505	380	2590	5
ST15	0	0	27	8	423	137	120	8
ST16	0	0	226	208	341	142	518	2
ST17a	0	0	365	86	1978	104	494	30
ST17b	0	0	457	75	510	363	407	23
ST17c	0	0	153	22	1204	141	602	5
ST18	0	0	500	8	0	43	61	16
ST19a	74	0	0	0	516	46	258	28
ST19b	60	0	176	3	208	30	463	54
ST20	0	0	0	0	0	0	4	10
ST21	0	0	651	0	2203	154	759	*
ST23a	0	0	14	0	20	4	0	39
ST23b	0	0	128	0	466	59	711	6
ST24a	29	0	26	0	0	0	0	51
ST24b	202	0	179	0	284	0	0	4
ST25a	375	263	82	38	741	174	9	1
ST25b	336	680	100	46	930	302	0	72

Site Code	Fish Biomass (g 100 m^{-2})							
	Trout	Salmon	Pike	Perch	Chub	Dace	Roach	Gudgeon
ST26a	217	136	0	0	56	1	0	11
ST26b	710	143	0	0	180	304	0	179
NW2	0	0	0	0	0	0	0	0
NW3	0	0	0	0	0	0	0	0
NW4	0	0	0	0	0	0	0	0
NW5	0	0	0	0	0	0	0	0
TRPB1	1172	199	0	0	0	0	0	0
TRPB1	2571	293	0	0	0	0	0	0
TRPB2	0	0	0	0	0	0	0	0
TRPB3	54	24	0	0	0	0	0	0
TRPB4	40	509	0	0	0	0	0	0
TRPB5	78	277	0	0	0	0	0	0
TRPB6	694	0	0	0	0	0	0	0
TRPB7	106	121	0	0	0	0	0	0
TRPB8	375	329	0	0	0	0	0	0
TRPB9	29	267	0	0	0	0	0	0
FRPB1	3	97	0	0	0	0	0	0
FRPB2	0	0	0	0	0	0	0	0