ASSESSMENT OF THE IMPACT OF DISCHARGES FROM SURFACE WATER SEWERS ON RECEIVING WATER QUALITY

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SYNOPSIS

A broad based approach has been used to assess the impact of discharges to rivers from surface water sewers, with the primary objective of determining whether such discharges have a measurable impact on water quality.

Three parameters, each reflecting the effects of intermittent pollution, were included in a field work programme of biological and chemical sampling and analysis which covered 47 sewer outfall sites. These parameters were the numbers and types of benthic macroinvertebrates upstream and downstream of the outfalls, the concentrations of metals in sediments, and the concentrations of metals in algae upstream and downstream of the outfalls.

Information on the sewered catchments was collected from Local Authorities and by observation at the time of sampling, and includes catchment areas, land uses, evidence of connection to the foul system, and receiving water quality classification. The methods used for site selection, sampling, laboratory analysis and data analysis are fully described, and the survey results presented.

Statistical and graphical analysis of the biological data, with the aid of BMWP scores, showed that there was a small but persistent fall in water quality downstream of the studied outfalls. Further analysis including the catchment information indicated that initial water quality, sewered catchment size, receiving stream size, and catchment land use were important factors in determining the impact.

Finally, the survey results were used to produce guidelines for the estimation of surface water sewer discharge impacts from knowledge of the catchment characteristics, so that planning authorities can consider water quality when new drainage systems are designed.

KEYWORDS RIVER QUALITY, URBAN RUNOFF, INTERMITTENT DISCHARGES, RECEIVING WATERS, MACROINVERTEBRATE INDICATORS.

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CHAPTER 1

INTRODUCTION

1.1 INTRODUCTION TO THESIS

A literature search and discussions with Severn-Trent Water and Birmingham City Council identified and confirmed the need for research into the effects of discharges from separate surface water drainage systems on receiving water quality. Planning authorities need to be able to predict effects on water quality so that adverse impacts can be minimised when systems are designed. This thesis describes the development, execution, results and conclusions of a research project designed to provide planners with an indication of the effects of discharges on receiving streams.

1.1.1 Research objectives

The major objectives of the research project were to

(1) Review and appraise published research on urban runoff quality, with emphasis on its impact on receiving water quality.

(2) Investigate, by a programme of field work, the impact of discharges from surface water systems on receiving water quality.

(3) Relate the findings from the field work programme to the characteristics of the catchments studied.

(4) If possible, use the results from (2) and (3) to formulate simple guidelines for estimating discharge impact from knowledge of catchment characteristics.

1.1.2 Arrangement of thesis

The remainder of this chapter provides the background to the research project and an introduction to UK river water quality, urban drainage systems, and water quality problems of urban areas.

In Chapter 2, published research into urban runoff and its effects on receiving water quality is reviewed, with references to literature studied at all stages of the project. The approaches adopted by other researchers were taken into account in the formulation of the project, which is described in Chapter 3, with emphasis on the rationale behind the decisions made and on the planning of the field work programme.

Field work techniques were assessed in a pilot study, reported in Chapter 4, which led to the selection of the methods for data collection and analysis described in Chapters 5 and 6.

Chapter 7 contains the results of the field survey, with statistical summaries, and these results are fully discussed and analysed in Chapters 8 to 10. In Chapter 11, the results of the data analysis are used to derive guidelines for impact estimation based on catchment characteristics, and additional work needed to improve and extend the guidelines is identified.

In the final chapter conclusions are drawn, and recommendations for further study are made.

1.2 BACKGROUND TO RESEARCH PROJECT

1.2.1 Previous work at Aston University

Aston University's involvement with urban runoff quality began in 1982, when Janet Wren took up a research studentship working on the separately sewered Chelmsley Wood housing estate in north-east Birmingham. The Chelmsley Wood site was equipped with sampling, flow and water quality monitoring instruments by Birmingham City Council, who had intermittently carried out experimental work there since the 1970s. Wren measured metal concentrations in runoff, roadside sediment, soil and the contents of deposit gauges, and reported her results in a PhD Thesis (Wren, 1986).

Interest stimulated by Wren's work led to the creation of two further research studentships, intended to commence in October 1985, to continue with urban runoff research. It was envisaged that the Chelmsley Wood site would continue to be central to the project, and a strong mathematical element was to lead to the development of an urban runoff quality model. The second research studentship was not taken up, however, resulting in the loss of most of the mathematical expertise, whilst the studentship which resulted in this thesis commenced in January 1986.

1.2.2 Project initiation

Two months were spent undertaking a preliminary literature review, and during this time it became apparent that although a considerable amount of data on runoff quality had been collected, its effects on receiving water quality had not been assessed. Discussions with Severn-Trent Water and Birmingham City Council confirmed that there was a need for information on such effects, as local authority and water quality planners had nothing on which to base their decisions when assessing drainage options.

This inevitably led to the decision to alter the initial direction of the project and study the effects of discharges on receiving streams. The preliminary literature review was then extended to include further detail on the effects of urban runoff discharges on receiving water quality. The entire literature review is reported in Chapter 2.

1.3 INTRODUCTION TO URBAN STORM DRAINAGE AND URBAN WATER QUALITY

1.3.1 Urban hydrology and the need for storm drainage systems

Urbanisation creates impervious surfaces which reduce infiltration thereby increasing the proportion of rainfall which appears as surface runoff. Development of urban areas also causes changes in microclimate, including an increase in rainfall event frequency and magnitude (Hall, 1984; Pope, 1980).

The hydrology of urban areas has been widely studied and reported (e.g Hall, 1984; Lazaro, 1979; Ward, 1975). A typical storm hydrograph for a rural area (Fig. 1.1) has a fairly steep rising limb and a gentle falling limb. As rain falls a proportion is intercepted by vegetation and the remainder reaches the ground where it is held in depression storage and begins to infiltrate the soil, forming the interflow and groundwater flow components of runoff. These subsurface flows take longer to contribute to streamflow than surface ones, so the falling limb of the hydrograph slopes more gently than the rising limb.

In urban areas interception losses are lower because there is less vegetation, and infiltration is greatly reduced by the presence of buildings and paved surfaces. A high proportion of the total rainfall input therefore becomes overland flow, altering the shape of the hydrograph to that illustrated in Fig. 1.2. The peak is higher and reached more quickly, and the falling

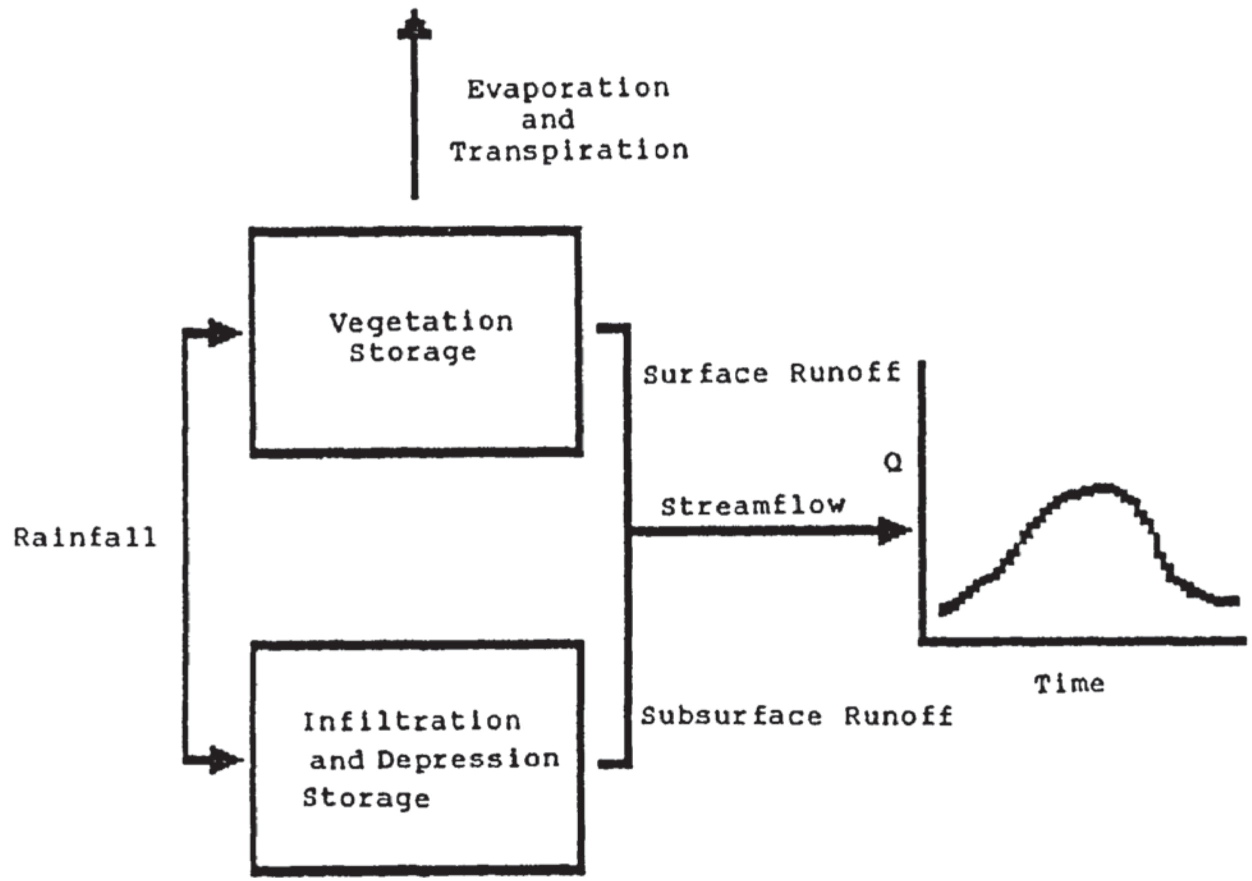

Fig. 1.1 Natural Catchment Before Urbanisation (after Lazaro, 1979)

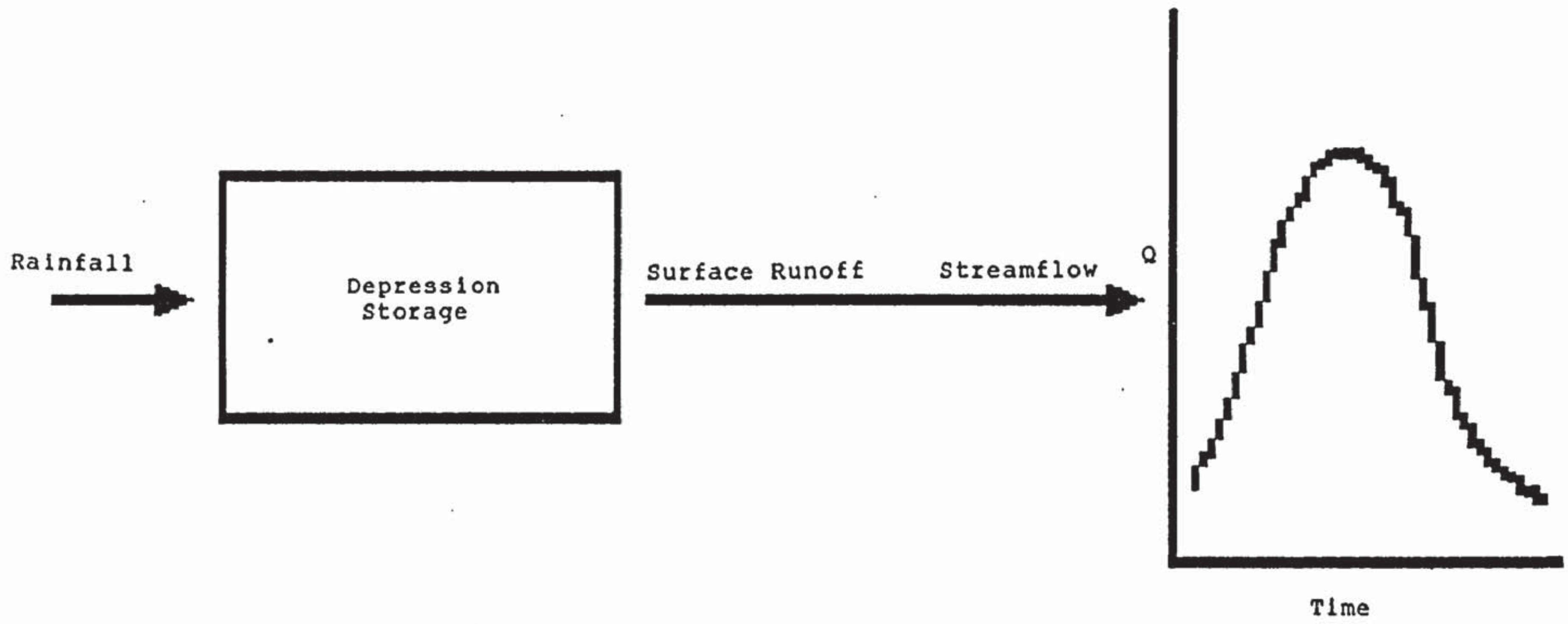

Fig. 1.2 100% Impervious Urban Catchment (after Lazaro, 1979)

limb is also more steeply sloped, since the subsurface flow contribution is greatly reduced.

To prevent flooding, runoff from an urban area must be rapidly removed by a storm drainage system, which generally consists of a network of pipes which carries the runoff directly to a convenient watercourse.

1.3.2 Types of storm drainage system

There are two basic types of storm drainage network:

(a) combined systems, in which foul sewage and urban runoff are conveyed in the same pipe; and

(b) separate systems, in which foul sewage is conveyed to a treatment plant in one pipe network and urban runoff is carried directly to a watercourse in another.

In practice, many systems exist which are a mixture of the two, such as 'partially separate' systems in which road and front roof runoff is collected in a separate pipe and rear roofs and paved areas behind houses drain to combined sewers.

Pre-1950 communities are usually sewered by combined or partially separate systems with separately sewered areas where recent development has taken place. Current practice for new developments is to construct separate systems.

During dry weather the foul sewage in combined systems is treated before discharge to a watercourse, but during a storm event the volume of foul sewage plus urban runoff (storm sewage) often exceeds the capacity of the treatment plant. Combined sewer overflows are therefore installed to limit the maximum flow in the sewer. Overflows usually discharge storm sewage directly to the nearest watercourse, without treatment. In separate systems, the foul sewage is always treated and the urban runoff is generally discharged to a convenient watercourse without treatment.

1.3.3 Storm drainage and water quality

The process of urbanisation generally leads to a deterioration in stream water quality as a result of the discharge of wastes to watercourses. Since infiltration is reduced by urbanisation, recharge to groundwater is low in urban areas and the flow in streams between storm events is therefore reduced. This in turn reduces the dilution available to those wastes discharged at times of low flow and thereby contributes further to the deterioration of urban stream water quality.

Separate sewer systems were introduced to avoid the need for combined sewer overflows which are believed to have a serious polluting effect and are often aesthetically offensive. Separate systems were installed in UK "New Towns" after 1950, and it was assumed that discharges to streams were not damaging to water quality. It is now recognised that urban runoff is contaminated by dissolved and suspended matter washed from roofs, streets, car parks, pavements and other surfaces. Reduced infiltration makes more water available for dilution of contaminants, but the rapid flows in urban areas increase the potential for erosion and transport of large particles.

Recent improvements in UK river water quality have drawn attention to the importance of discharges from combined sewer overflows and surface water sewers. The Department of the Environment 1985 survey of river quality (Department of the Environment, 1986) showed that although 90% of river reaches in England and Wales were of satisfactory quality, approximately

3700km of rivers are of poor quality, and these are predominantly in densely populated urban areas. Combined sewer overflows have been identified as one of the major factors limiting the quality of river water in many areas (Clifforde et al, 1986), but the effects of discharges from surface water systems, although recognised, have received less attention. The pollution threat of discharges from surface systems is likely to be less than that from combined sewer overflows, but is probably still significant.

As river quality continues to improve, the attention paid to surface water sewer discharges is likely to increase in importance. Concern for river quality in the last 20 years has resulted in a considerable amount of research being undertaken into urban runoff quality, control, treatment, and prediction. The literature on all aspects of urban runoff quality has been studied in order to gain insight into the potential environmental damage caused by its discharge and to identify specific areas in which research is needed. This literature is reviewed in Chapter 2.

CHAPTER 2

URBAN RUNOFF QUALITY AND EFFECTS ON RECEIVING WATERS

This chapter contains an account of the literature reviewed at all stages of the research project, under the following headings:

Urban runoff quality
Sources and nature of contaminants
Urban runoff management
Effects on receiving waters

Published research was identified by on-line database searches (Aqualine and Water Resources Abstracts), direct requests to the US Environmental Protection Agency (EPA), and regular browsing through journals in specialist libraries. Current research was identified by attending conferences, meetings and seminars.

Because of the large volume of published research, the following review was restricted to UK, European and US studies, with emphasis on work carried out in the UK.

2.1 URBAN RUNOFF QUALITY

Most urban runoff quality research, especially in the UK, has been carried out at the single catchment or subcatchment level, often concentrating on a single aspect of the subject or a specific group of pollutants. This work has been largely unco-ordinated and its site-specific nature makes comparisons between catchments difficult. Increased awareness in recent years of the potential polluting ability of urban runoff has led to better co-ordination of research effort, and the data requirements of water quality simulation models have resulted in more information being collected. The majority of this work has been in the US, however, with few catchment studies in the UK, and although work is continuing there is still a lack of reliable UK urban runoff quality data.

Many factors have been quoted as influencing the quality of runoff discharged to watercourses, including the physical characteristics of the catchment, land use within the catchment, the nature and extent of contaminants present, climatic and seasonal variability, the length of antecedent dry weather period, roadside gully pots, and the erosion of in-pipe deposits, and these are considered below.

2.1.1 UK urban runoff quality

One of the earliest studies of urban runoff quality was carried out at Oxhey, a separately sewered housing estate near Watford, in the mid-1950s (Wilkinson, 1956). Wilkinson reported Biochemical Oxygen Demand (BOD) values up to 100 mg/l and suspended solids contents in excess of 2000 mg/l.

Following Wilkinson's work, few runoff quality studies were conducted in the UK until the 1970s, since which time experimental work has been carried out at a number of locations. In addition to Oxhey, where further work has recently been carried out (Harrop, 1984), monitored UK catchments include Grahams Park, Hendon (Ellis, 1977); Clifton Grove, Nottingham (Fletcher et al, 1978; Pratt and Fulcher, 1987); Welbeck, Nottingham and Rise Park, Nottingham (Tucker and Mortimer, 1978); Chelmsley Wood, Birmingham (data held by WRc but source unknown; Wren, 1986); Shephall, Stevenage, and Stevenage New Town (Mance and Harman, 1978). Two further studies have examined motorway runoff quality - on the Aston Expressway, Birmingham (Hedley and Lockley, 1975); and the M1 at Toddington (Pope et al, 1978); and runoff from the urban Basils Road in Stevenage has been monitored

(Mance, 1981).

The physical characteristics of these catchments were summarised by Mance (1981) who noted their similarity in terms of age, rainfall and land use. A brief summary of the catchments is given in Table 2.1. Only the larger catchments include industrial areas, and most of the areas are recent suburban, consisting of houses, shops and schools.

TABLE 2.1 CHARACTERISTICS OF MONITORED UK CATCHMENTS

Catchment	Area ha	Impervious %	Description
Oxhey	247	20	Recent residential
Grahams Park	350	39	Mixed housing, light industry, and motorway
Clifton Grove	10.6	52	Recent residential
Welbeck	16	35	Recent residentail (1960)
Rise Park	62	31	Recent residential, some construction in progress
Chelmsley Wood	107	35	Recent residential
Shephall	155	21	Recent residential
Stevenage New Town	2500	30	Mixed housing, industry, and construction
Aston Expressway	3.6	100	Urban motorway
M1 Toddington	0.08	100	Motorway
Basils Road	0.38	100	Mature residential road

Mance also experienced difficulty in making quality comparisons between catchments because of the different determinands measured, with suspended solids being the only determinand common to all the studies. The most commonly quoted quality parameters are listed in Table 2.2 for the catchments where data are available. Also given in Table 2.2 are typical treated sewage quality (Tebbutt, 1971) and the EC guide level concentrations for surface water intended for abstraction for drinking (with simple physical treatment) and for water supporting coarse freshwater fish (Council of the European Communities, 1975 and 1978; Gardiner and Mance, 1984). These standards are included for comparison purposes only, as they are unlikely to be applicable to many streams receiving urban runoff discharges. Many determinands not listed in Table 2.2 have been studied, including ammonia, nitrite, nitrate, chloride, phosphates, pH, zinc, copper, lead, nickel, chromium, and cadmium; but data for these are not available for many catchments.

TABLE 2.2 SELECTED QUALITY PARAMETERS FOR UK CATCHMENTS

Catchment	TSS		TS		BOD		COD	
	mean	max	mean	max	mean	max	mean	max
Oxhey (Wilkinson)	194	2045	-	-	7	100	-	-
Oxhey (Harrop)	101	939	271	1474	-	-	-	-
Grahams Park	546	2688	978	-	22	-	265	-
Clifton Grove	21	213	418	17688	7	46	39	163
Chelmsley Wood	130	-	565	-	14	-	134	-
Shephall	112	1386	364	2623	-	-	-	-
Aston Expressway	134	2599	-	-	4	65	-	-
Basils Road	28	-	-	-	-	-	-	-
Abstraction for drinking water	25		-		3		-	
Coarse freshwater fish	25		-		6		-	
Treated sewage	17		-		12		-	

All concentrations in mg/l
TSS=total suspended solids; TS=total solids

There is great variation in the concentrations of pollutants in urban runoff, even for the few parameters and similar catchments in Table 2.2. It is clear from comparison with standards, however, that runoff is likely to have a detrimental effect on the receiving water.

Studies of highway drainage have indicated that water draining from residential roads is not generally of poorer quality than water draining from other parts of residential catchments. The figures reported by the Hendon and Aston Expressway studies, however, indicate that pollutants in motorway runoff are present in higher concentrations than other catchments. Motor vehicles as a source of pollutants are discussed in section 2.2.

Mance and Harman sampled four snow-melt events and concluded that snow-melt runoff quality was of poorer quality than rainfall runoff. With the exception of ammonia, which reached maximum concentrations in the dry summer months, pollutants were present in higher concentrations in snow-melt runoff than in rainwater runoff. The reasons suggested for this were heavy applications of road salt, less efficient utilisation of petrol by motor vehicles, and contact between snow and vehicle bodywork. Pope (1980) also observed that the application of road salt may have the indirect effect of elevating metals levels in runoff by increasing the ionic strength of the water and enhancing its ion exchange capacity.

Most of the UK studies have examined variation in runoff quality with flow and time since the start of the rainfall event, and many have observed a first flush of pollutants at the start of runoff. The first flush has been defined in various ways, but can be simply described as the occurrence of peak pollutant concentrations before the peak discharge occurs (Fig. 2.1). At Oxhey, Wilkinson observed a first flush for BOD and permanganate value (PV) with the mass of these in the first 55,000 gallons discharged increasing with the length of the antecedent dry weather period up to 10 days, after which no further increase occurred. The Oxhey first flush was also considered on a time basis, and between 43% and 50% of the pollutants were discharged by 29% of the water in the first 30 minutes of flow.

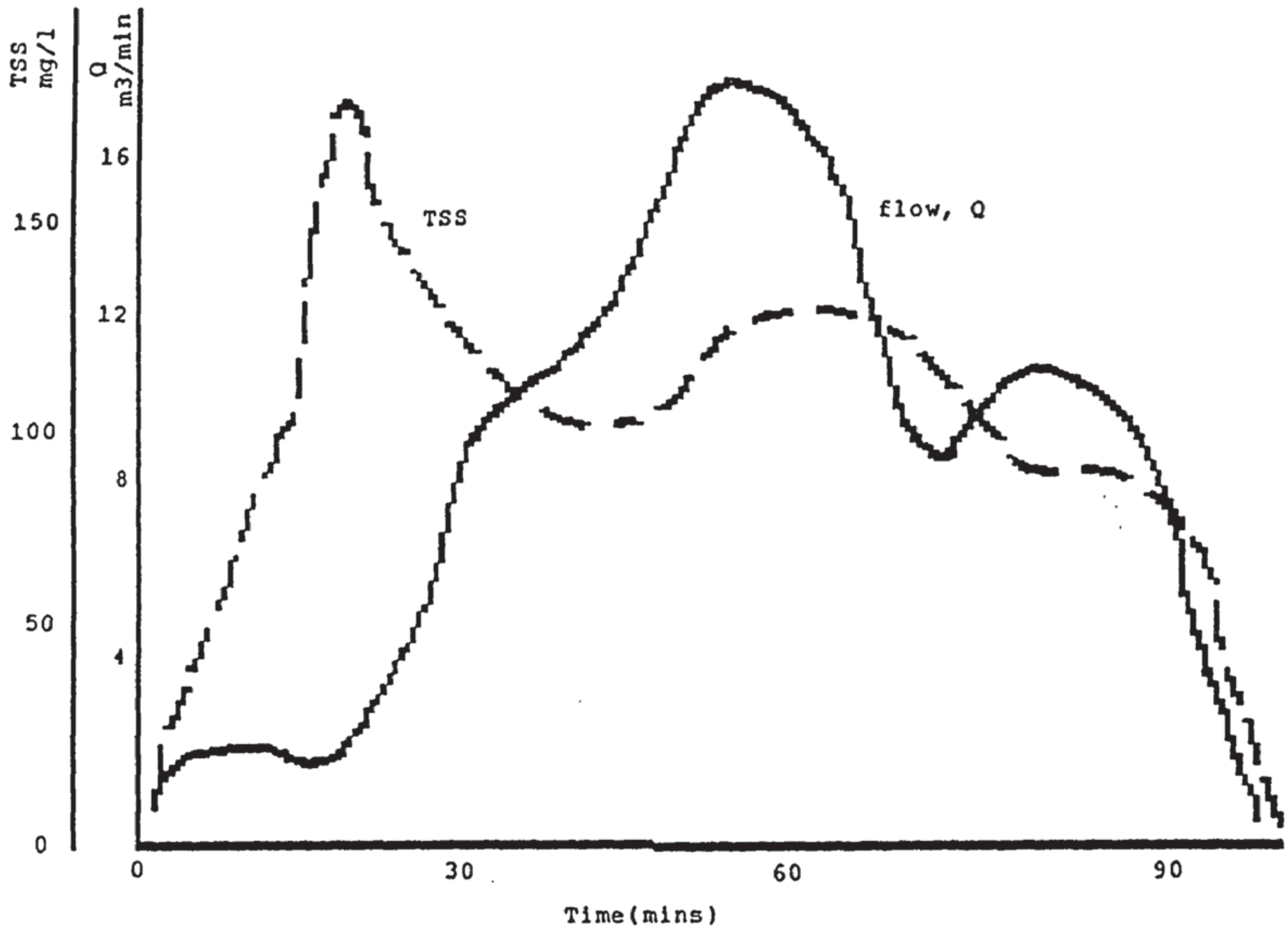

Fig 2.1 Suspended Solids Pollutograph with First Flush Effect

The first flush of pollutants has also been reported by Tucker and Mortimer (1978), Ellis (1977), Mance and Harman (1978), Mance (1981), Wren et al (1984) and Harrop (1984). Mance reported that for the majority of storms at Shephall, the maximum pollutant concentrations occurred in the first 20 minutes of flow, and were usually before or coincidental with the peak discharge. This effect was most marked for soluble pollutants. Using a definition of first flush based on cumulative mass pollutant flows suggested by Tucker and Mortimer, Mance also demonstrated source limitation effects for TSS, chloride and copper. Ellis (1976) observed a double peak in TSS concentrations, the first occurring early in the hydrograph, and the second on the back of the falling limb of the hydrograph.

The first flush of pollutants discharged from separate systems has been attributed to washing of the catchment surface, erosion of in-pipe deposits, and the flushing of contaminants from gully pots. All of these factors probably operate with catchment, storm, and seasonal influences, to determine the magnitude of the first flush; which accounts for the wide variation reported. The double peak of solids reported by Ellis is best explained as an initial flushing of sediment from within the system, followed by new material from the surface at the time of concentration of the catchment. Fletcher et al (1978) cite gully pots as a major contributing factor to first flush events due to the poor quality of gully liquor, particularly during the summer when anaerobic conditions are quickly reached.

The length of the antecedent dry period has been linked to the magnitude of the first flush and to runoff quality generally. It is reasonable to assume that during dry weather periods pollutants accumulate on catchment surfaces, and that the longer the dry period, the greater the accumulation. Wilkinson confirmed that this was true for BOD and PV, but not for TSS, and that the levels of pollutants stopped increasing after 8 to 10 days. Mance confirms that for consecutive storms, the quality of runoff is higher for the second and subsequent storms. Ellis (1986) found that flow and event duration controlled removal of metals from street surfaces, and that antecedent dry period had little effect. The subject of pollutant buildup and removal has received more attention in the US, and is further discussed in section 2.2.

2.1.2 European and US urban runoff quality

European runoff quality has been reported for Norway (Lindholm and Balmer, 1978; Reinertsen, 1982); Sweden (Soderlund and Lehtinen, 1972; Malmquist and Svensson, 1977); Germany (Goettle, 1978); France (Hemain, 1982; Deutsch and Desbordes, 1982; Deutsch and Hemain, 1984); and The Netherlands (van Dam et al, 1986). The earlier of these studies were included in Mance's 1981 review and the differences between European and UK runoff quality were no greater than those within the UK data set.

Reinertsen investigated runoff quality from small areas of streets, parking lots and sidewalks, and found that TSS, COD and lead concentrations were higher in runoff from heavily trafficked areas than residential areas. Maximum concentrations measured were TSS 4990 mg/l, COD 870 mg/l, and lead 3.09 mg/l. No relationship was found between pollutant concentrations and discharge.

Four French residential catchments were monitored for one year: two in the Paris area, and two in Aix en Provence. Mean TSS concentrations varied between the catchments from 191 to 473 mg/l; COD from 77 to 278 mg/l; and BOD from 12 to 45 mg/l. Maximum observed values were TSS 804 mg/l, COD 780 mg/l, and BOD 140 mg/l. The higher population density catchments of Aix en Provence produced consistently poorer quality runoff than the single family housing areas near Paris.

In the Netherlands, rainfall and runoff from roofs, motorway, and a subcatchment of the New Town of Lelystad have been monitored for quality. Median concentrations in runoff for TSS, COD and BOD were 45, 30 and 3.7 mg/l respectively, with higher values for motorway runoff. For many constituents, especially lead and zinc, concentrations in rainfall were higher than in roof runoff. This was attributed to their binding to particles and algae on the roof, and confirmed by the observation that they were washed off during high intensity storms.

The great majority of runoff quality data has been collected in the US. Early studies included those reported by Palmer, (1963), and Weibel et al, (1964). Palmer reported TSS concentrations of over 200 mg/l and total coliforms of 430,000 (MPN/100ml); and Weibel et al found maximum TSS, COD and BOD concentrations of 1200, 610 and 84 mg/l respectively in runoff from a residential and light commercial area of Cincinnati. Weibel also studied the bacterial quality of runoff, and found total coliforms exceeding 2900 colonies per 100 ml in 90% of samples. The incidence of micro-organisms in runoff has recently been reviewed (Payne and Moys, 1989) with the general observation that there is very little published information on the bacterial quality of urban runoff. The available information is all from the US and Canada, where studies have reported that total coliforms densities in runoff are about 10% of those found in combined systems.

Bradford (1977) produced a statistical summary of selected US data published up to 1972, and identified a number of determinands and catchment characteristics which had been considered relevant in various studies (Table 2.3). Although the list of parameters included only the most commonly quoted, many of these were omitted from subsequent calculations because of insufficient data.

TABLE 2.3 PARAMETERS IDENTIFIED IN EARLY US URBAN RUNOFF STUDIES

Determinands	Catchment characteristics
Solids loading rate	Climate
BOD	Season*
COD	Land use
Ortho-phosphate	Population density*
Total phosphorus	Age of catchment*
Nitrate	Impervious fraction*
Ammonia	Average daily traffic
Organic nitrogen	Street surface material
Cadmium	Landscaping
Chromium	Fraction of area as road*
Copper*	Days since last rain*
Iron	Days since streets cleaned*
Lead	Method of street cleaning*
Nickel	
Strontium	
Zinc	
Total coliforms	
Faecal coliforms	
Pesticides*	

* denotes insufficient data for consideration

Further studies were carried out in the US in the 1970s, including those by Colston (1974) and Whipple et al (1978). The lack of co-ordination of urban runoff quality research in the US eventually led to the Nationwide Urban Runoff Program (NURP), made up of 28 individual projects which monitored runoff quality from 1978 to 1983. NURP projects were co-ordinated by the EPA, with the principal objective of developing an information base to support water quality planning efforts. In addition to final reports from each individual project and papers published as a result of the program (e.g Cole et al, 1984; Myers et al, 1982), NURP results have been summarised and analysed in a series of EPA reports (US EPA 1983a, 1983b and 1983c). One of the NURP objectives was to characterise urban runoff quality, and determine differences or similarities between different urban locations. The other NURP objectives were concerned with receiving water quality and are discussed in section 2.4.

NURP selected the following list of 'standard pollutants' which were measured in most of the individual studies:

Total suspended solids	TSS
Biochemical oxygen demand	BOD
Chemical oxygen demand	COD
Total phosphorus as P	TP
Soluble phosphorus as P	SP
Total Kjeldahl nitrogen as N	TKN
Nitrate and nitrite as N	TON
Total copper	Cu
Total lead	Pb
Total zinc	Zn

In addition to these was a list of 'priority pollutants' covering 129 specific compounds or classes of compound in 10 groups:

I	Pesticides
II	Metals and inorganics
III	PCBs and related compounds
IV	Halogenated aliphatics
V	Ethers
VI	Monocyclic aromatics excluding VI and VIII
VII	Phenols and cresols
VIII	Phthalate esters
IX	PAHs
X	Nitrosamines and other N compounds

Approximately two-thirds of projects participated in the priority pollutant program. A vast amount of data was collected and examined for various relationships including the effects of geographic location, land use, runoff volume, catchment slope, population density, and soil type on runoff quality. Most of this analysis was performed on site median EMC data (EMC = event mean concentration, as total constituent mass discharge/total runoff volume) and within-event quality variation was not considered. No significant correlation was found between runoff quality and any of the site or event characteristics. The NURP results were pooled to produce typical concentrations in urban runoff of the standard pollutants (Table 2.4).

TABLE 2.4 WATER QUALITY CHARACTERISTICS OF URBAN RUNOFF

Constituent	Event to Event Variability in EMCs (Coef Var)	Site Median EMC	
		For Median Urban Site	For 90th Percentile Urban Site
TSS (mg/l)	1-2	100	300
BOD (mg/l)	0.5-1.0	9	15
COD (mg/l)	0.5-1.0	65	140
Tot. P (mg/l)	0.5-1.0	0.33	0.70
Sol. P (mg/l)	0.5-1.0	0.12	0.21
TKN (mg/l)	0.5-1.0	1.50	3.30
TON (mg/l)	0.5-1.0	0.68	1.75
Tot. Cu (ug/l)	0.5-1.0	34	93
Tot. Pb (ug/l)	0.5-1.0	144	350
Tot. Zn (ug/l)	0.5-1.0	160	500

The most frequently detected priority pollutants were toxic metals including copper, lead, zinc, arsenic, chromium, cadmium and nickel. Cyanides were also frequently detected. One of the NURP conclusions was that heavy metals were the most prevalent pollutants in runoff. Mean annual loads were also calculated, and compared with others cited in the literature for different land uses and with those from a well-run secondary sewage treatment plant. The following general conclusions were reached:

a) Annual TSS loadings from urban runoff are approximately one order of magnitude greater than those from a sewage treatment works

b) BOD loadings are of the same order of magnitude

c) Total P loadings from urban runoff are approximately one order of magnitude lower than those from a sewage treatment works

d) Pollutant loads compared well with other cited in the literature.

These conclusions support earlier studies which have compared runoff quality to secondary sewage effluent, and indicate that better management of runoff is necessary if water quality objectives are to be met.

Various control measures have been proposed to reduce the pollution load from separate drainage systems, and a considerable amount of work has been carried out on the source and precise nature of catchment surface pollutants, with the aim of achieving better control through better understanding of the processes involved in pollutant accumulation, washoff and transport. Studies of the source and nature of pollutants in urban runoff are reviewed in section 2.2.

2.2 SOURCES AND NATURE OF CONTAMINANTS

The complete qualitative and quantitative evaluation of all pollution sources in an urban catchment is not feasible. Extensive quantitative work has been carried out, however, to establish mechanisms of pollutant buildup and washoff from road surfaces and to characterise roadside sediments in terms of their polluting strength.

The major sources of pollutants in urban runoff have recently been listed by Novotny et al (1985) as follows:

Wet and dry atmospheric deposition
Street refuse deposition including litter, street dirt, vegetation and organic residues
Traffic emissions and impact
Urban erosion
Road de-icing

These inputs and associated outputs from a road surface are illustrated in Fig. 2.2. and are discussed in more detail below. Additional sources of pollutants discharged to watercourses from urban drainage systems are unintentional or intentional chemical spillages and connections to the foul sewerage system,

2.2.1 Wet and dry atmospheric deposition

Sources of atmospheric pollutants which can be washed out by rainfall or deposited in dry weather are numerous. Following their emission from factories, motor vehicles, or exposed land surfaces, pollutants are dispersed in the atmosphere where they may also undergo chemical reactions before deposition.

Rainfall in urban areas with sulphur dioxide and nitrogen oxides sources is commonly acidic, and this may aid washout of metallic particles and cause corrosion of vehicles and street furniture (Hedley and Lockley, 1975; Fletcher et al, 1978). Randall et al (1978), Goettle (1978) and Malmquist (1978) found that a significant proportion of loads for some pollutants, particularly nitrogen compounds, can be attributed to wet deposition. This was confirmed by Ng (1987) who found that most of the dissolved copper and nickel in runoff could be attributed to their presence in rainwater.

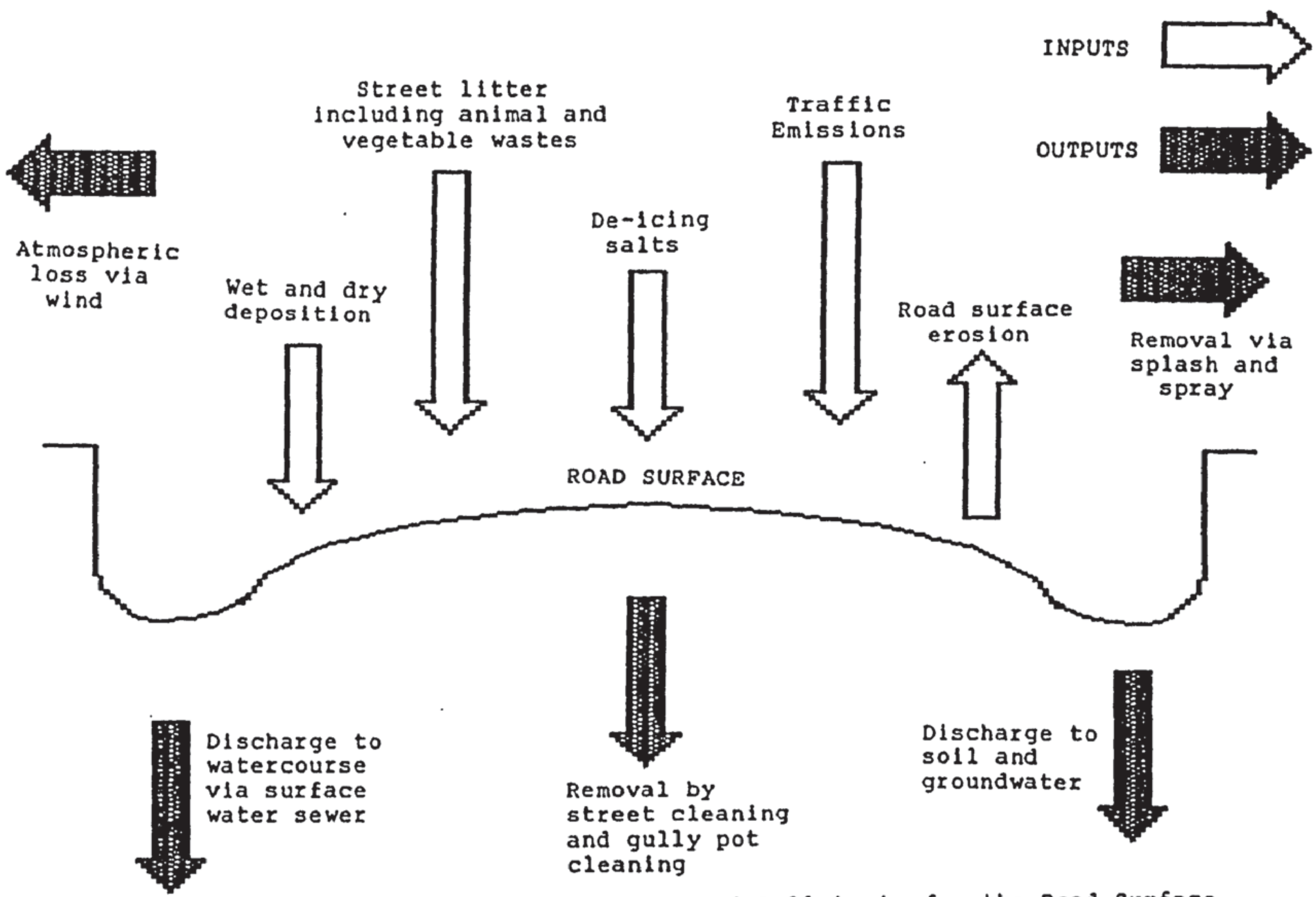

Fig. 2.2 Input and Output of Pollutants for the Road Surface
(adapted from Pope, 1980)

The contribution of ammonia from rainwater in reported studies consistently exceeds the reported outputs. (Mance, 1981; Ng, 1987). No satisfactory explanation for this has been found in the literature but it appears that some chemical change, presumably to nitrite and nitrate, takes place in the catchment, between rainfall and runoff sampling points.

Several attempts have been made to quantify the atmospheric deposition contribution to pollutants in runoff, and Table 2.5 lists contributions determined by two studies in urban areas, for dustfall (Kettner, 1974) and total deposition (Mance and Harman, 1978) respectively.

TABLE 2.5 CONTRIBUTION OF ATMOSPHERIC DEPOSITION TO URBAN RUNOFF POLLUTANTS

Pollutant	% Contribution	
	Dry	Wet+Dry
Total suspended solids		23
Volatile suspended solids	30 - 40	
Soluble solids	-	30
Total solids	-	29
Chloride	1 - 5	28
Nitrate - N	0.5 - 1	68
Nitrite - N	-	50
Ammonia - N	0.5 - 1	417
Copper (as CuO for dustfall)	0.01 - 0.1	96
Zinc (as ZnO for dustfall)	0.01 - 0.5	38
Lead (as PbO for dustfall)	0.05 - 0.2	54

Other estimates of atmospheric deposition pollutant inputs include those of Ellis et al (1986), who report that aerial deposition is the primary source of metals in a road drainage system on a housing estate in London; Heaney and Sullivan (1971), who estimated that 70% of all street surface material originated from atmospheric depostion; and Sartor and Boyd (1972) who found that quantities of metals and nutrients on street surfaces were significantly influenced by dustfall.

A further observation, reported by Randall et al (1978) was that pollutants were washed out during the early stages of a rainfall event, contributing to the first flush, and that the resulting input to the catchment surface was thus independent of rainfall intensity and duration. Wet deposition was also found to be fairly uniform across the catchment studied, indicating that dispersion in the atmosphere was operating, as the sources of pollutants were more specific.

Pollutant quantities in atmospheric deposits, particularly those washed out by rainfall, are of sufficient magnitude to affect the quality of urban runoff. This is especially true for ammonia and other nitrogen compounds, and metals, particularly copper.

2.2.2 Street refuse deposition

Street refuse refers to particles too large to be classed as dust and includes litter such as food, cans, paper and plastics; fallen leaves and other plant litter; and animal faeces and carcasses. The type and amount of street refuse present is very variable, depending, amongst other factors, on land use, season, and the level of human activity in the area.

Plant litter includes leaves, grass cuttings, fruit, twigs and seeds; and may have been intentionally dumped on roads or blown by the wind from gardens and other planted areas. Garden chemicals including pesticides may be associated with plant debris.

Organic matter increases the BOD and COD of runoff, and animal faeces and remains are also a possible source of pathogens (Water Research Centre, 1977). The general cleanliness of an area, and in particular the presence of uncovered rubbish, was concluded by Olivieri et al (1989) to have a significant effect on the bacteriological quality of runoff.

Large quantities of plant debris, such as generated by autumn leaf-fall, contribute to high organic loadings to gully-pots and can cause blockages as well as high oxygen demands. Pratt and Adams (1981) have demonstrated seasonal variation in levels of organic matter, with a marked increase in autumn. Sartor and Boyd (1972) state that accumulations of plant material in gully pots can exert a long-term oxygen demand as well as the more commonly measured 5-day BOD.

Large floatable litter such as paper packages can result in blockages of the drainage system, and if transported to watercourses are an aesthetic nuisance.

Although the organic fraction of litter is generally low relative to inorganic matter (Sartor and Boyd, 1972), it is likely to be significant to water quality in terms of oxygen demand and bacterial inputs.

2.2.3 Traffic emissions

Pollutants originating from motor vehicles have been extensively studied in the UK, US and Europe. Hedley and Lockley (1975) in the UK and Sartor and Boyd (1972) in the US have produced similar classifications of vehicular pollutant inputs, and these are summarised as follows:

a) Leakage of fuel, lubricants and other fluids

b) Exhaust emissions

c) Fine particulates worn from tyres, clutches and brakes

d) Dirt, rust and decomposing coatings from vehicles

e) Vehicle components broken by impact (glass and metals)

Metal pollutants, from exhausts and from wear and corrosion of vehicle components, are generally considered the most important inputs; and of these lead has received the most attention.

Lead is added to petrol as tetraethyl and tetramethyl lead, and is emitted from exhaust systems primarily in particulate inorganic compounds (Laxen and Harrison, 1977). The addition of ethylene dibromide and ethylene dichloride to petrol as lead scavengers results in the presence of lead halides and oxyhalides in vehicle exhaust, with lesser amounts of lead phosphates and sulphates. Other noxious pollutants in vehicle exhaust include carbon monoxide, nitrogen oxides, sulphur oxides and hydrocarbons.

Part of the lead emitted by vehicles is deposited on the road surface where it becomes available for washoff by rainfall. Several studies have demonstrated that the levels of particulate lead decrease with distance from the road (Colwill et al, 1974; Deroanne-Bauvin et al, 1987; Warren and Birch, 1987), as the majority of lead is deposited on or near the road itself. This is undoubtedly the cause of the elevated lead levels observed in runoff from major roads and motorways.

Release of pollutants as a result of tyre wear provides a second significant input which has been reported by Cadle and Williams (1978) and Christensen and Guinn (1979). Cadle and Williams reported that hydrocarbons are emitted from tyres under normal driving conditions, and Christensen and Guinn found that zinc, used in the manufacture of tyres, was deposited on road surfaces at a rate of 0.003g Zn / vehicle km.

Other reported pollutants of vehicular origin are metals from engines and car bodies, including cadmium, chromium, nickel, zinc, copper, and manganese; all of which have been found in road surface sediments.

2.2.4 Other pollutant sources

Pollutant sources not considered above are urban surfaces themselves, de-icing salts, gully pots, spillages and the foul sewerage system.

Erosion of solids from urban surfaces is only significant as a source of pollutants where there are construction sites or open land, or when rainfall is very heavy and vegetation is unable to protect soil (Novotny et al, 1985).

Mance and Harman (1978) found that roof runoff was cleaner than road runoff and had a diluting effect rather than a polluting one. Studies in Sweden, however, where copper and zinc are used in roofing and guttering materials, have demonstrated that increased levels of these metals occur in runoff as a result of corrosion (Malmquist and Svensson, 1977). Other urban surfaces which contribute pollutants as a result of corrosion include railings and street furniture in general, but inputs from these are probably small.

De-icing activities have been cited by several studies as a source of contaminants (Hedley and Lockley, 1975; Fletcher et al, 1978; Pope, 1980). De-icing salts contribute directly to suspended solids, chloride, nickel and chromium loads. Application of salt also has indirect effects, as it stimulates corrosion and can lead to increased mobilisation of mercury (Pope, 1980). Ellis (1976) describes how de-icing salts in receiving waters can cause release into solution of metals held in benthal sludges. Pollutants in de-icing salts are also partly responsible for the lower quality of snowmelt runoff compared with rainfall runoff. The use of urea in some de-icing compounds results in elevated nutrient levels in runoff, which can cause problems of excessive algal and weed growth in the receiving water.

Gully pots as a source of pollution has been the subject of several studies on the Clifton Grove catchment in Nottingham (Fletcher et al, 1978; Pratt et al, 1986) and a smaller scale study in Stevenage (Mance and Harman, 1978). Gully pots act as inlets to storm drainage sytems from roads, and are intended to remove solids which might otherwise cause blockages. The UK studies have concluded that whilst gully pots efficiently remove large solids, they represent a significant reservoir of poor quality water which contributes considerably to the quality of water entering surface water sewers. This is particularly true in summer dry periods, which result in increased BOD, COD, nitrate, and ammonia concentrations in gully liquor. Following a rainfall event, the gully liquor quickly becomes anoxic, and resulting bacterial scums and soluble waste products contribute to the first flush of pollutants in the sewer.

The effect on water quality of chemicals entering surface water systems can be devastating. Sources of chemicals range from domestic - detergents and bleach, garden pesticides, fertilisers etc. - to hazardous load transporters and major industrial spills. Many domestic spillages are due to ignorance of the nature of the drainage system, and Hall and Ellis (1985) have suggested that public awareness of problems caused by the use of household chemicals could be increased by advertising campaigns. Pope (1980) describes the more serious potential problems of hazardous load transportation. Although chemical pollution of watercourses as a direct result of spillages can cause massive fish kills and difficulties for downstream users, drainage of spilled toxic materials to sewage works may be even more serious; as physical and chemical disruption of treatment processes results in untreated sewage being discharged in addition to the spilled chemical.

A final source of pollutants in surface water sewers is the foul sewerage system. Although it is widely acknowledged that the majority of separate surface water systems contain foul sewage, this has not been the subject of any known water quality study to date. Connections to the foul system may be deliberate or accidental, or indirect contamination may be present as a result of damage to pipes. Small household plumbing operations which are carried out by inexperienced workers are the likely origin of many small illegal connections. A particularly common observation is that washing machine outlets drain to surface rather than foul sewers.

2.2.5 Characteristics of surface particulates

Pollutants, largely in particulate form, originating from the various above-ground sources described accumulate on catchment surfaces where they are available for removal by rainfall or street cleaning. Their eventual effects on receiving waters depend on the quantities in which they are present, their chemical and bacterial quality, and the ease with which they can be transported by rainfall and overland flows.

Several detailed studies have been undertaken to investigate these phenomena, largely in relation to water quality enhancement via the use of mathematical models and improved street cleaning practices. In the UK, much experimental work has been carried out at Middlesex Polytechnic (e.g. Ellis, 1976, 1977 and 1979; Ellis and Revitt, 1982; Ellis et al, 1986). An extensive study has been carried out in the US by Sartor and Boyd (1972; Sartor et al, 1974; Sartor and Gaboury, 1984).

Sartor and Boyd found material on streets ranging in size from

3000 to 75 microns or less. The bulk of the material was inorganic and in the sand and silt size ranges. Ellis reported that the inorganic mineral fraction made up 50 to 80% of the sample weight and included substantial amounts of brick, glass and aggregates. The organic fraction was variable but could account for up to a third of the total weight. Pratt and Adams (1982) recorded higher proportions of organic material in sediments containing fallen leaves in autumn. Ellis noted that between 15 and 70% of the organic material was associated with small particles of less than 0.06mm.

UK and US investigations have reported that a large proportion of the contaminants in runoff is associated with fine solids. Ellis, and Sartor and Boyd have both demonstrated that although fine solids account for a small proportion of the total weight of solids on street surfaces, they account for a large proportion of many contaminants. Ellis found between 40 and 90% of all contaminants associated with particles below 0.2mm. Particles below 0.06mm in size accounted for 25% of the oxygen demand, 30 to 50% of algal nutrients, 30% of metals, 50 to 60% of grease and rubber, and 10% of the total coliforms; although only 4 to 8% of particles belonged to this size range. Similar figures have been reported by Sartor and Boyd, with particles below 43 microns accounting for 25% of the oxygen demand, and over 50% of metals.

Sartor and Boyd also studied the accumulation of surface sediments. They observed that the particulate pollutant load increased rapidly following a washoff or street cleaning event, and that after several days the buildup rate fell. This was attributed to the influence of turbulence generated by wind and motor vehicles. The quantity of pollutants present was observed to vary with road surface type and condition, with asphalted streets exhibiting 65% greater loadings than concrete streets and poor condition surfaces exhibiting loadings over twice those of new surfaces.

The distribution of solids across road surfaces was found to be very uneven. Sartor and Boyd found 78% of solids within 6" of the kerb, and Ellis found over 80% within a metre of the kerb. Both concluded that the kerb acted as a protective barrier for particle accumulation, and that where kerbs were low or absent the phenomenon was not apparent.

The association of the bulk of pollutants in road surface sediments with the fine size fraction is of particular relevance to street sweeping and other control measures which have been suggested to improve the quality of runoff and reduce the pollution load to receiving waters. Control and treatment practices are discussed in section 2.3.

2.3 URBAN RUNOFF POLLUTION MANAGEMENT

The problems of urban runoff management are those of flooding caused by hydraulic overloading of sewerage systems, and pollution caused by combined sewer overflows and surface water discharges. The practices discussed in this section relate to those used to reduce pollution from separate surface water systems.

Various controls have been suggested to reduce the pollution load of surface runoff, and these have been broadly categorised by Harremoes (1982) into source controls, street cleaning and runoff treatment. The theory behind source controls is to limit the supply of pollutants. This can be achieved by flow attenuation, erosion control, restrictions on chemical use, and improved sanitation practices (Field and Lager, 1975). Source controls which do not involve structural measures are generally preferred for both economic and aesthetic reasons. Where treatment is necessary, Zanoni (1986) has demonstrated that the characteristics of solids settled from runoff differ from those of other sewage sludges, and that treatment methods currently used for sewage may not be applicable to urban runoff.

Wanielista and Yousef (1986) have produced a more detailed breakdown of runoff management practices with clear definitions:

1. On-line retention/detention - structures in which runoff is stored for disposal primarily to surface waters and to a lesser extent by evaporation, and infiltration.

2. Off-line retention - structures which include diversion and infiltration basins in which the first flush volume of runoff is disposed of primarily by infiltration, but also by evaporation, with no surface discharge from the infiltration basin.

3. Street cleaning - cleaning of streets by both vacuum and brush machines to dispose of some materials before removal to surface waters.

4. Catchbasin (gully pot) cleaning - mechanisms to dispose of materials stored in the bottom of a catchbasin before the materials are removed by runoff waters.

5. Erosion control - vegetation, chemicals or structures that prevent soil disposal from a site.

6. Roof and parking lot infiltration - a specific application of off-line retention to dispose of the first volume of runoff primarily by infiltration.

In addition to studies evaluating the performance of these control measures, much effort has been spent on the development of simulation models with the capability to analyse flows and pollution loads throughout sewerage systems. Such models provide the key to economical upgrading of existing systems and better design of new systems, and can be used in conjunction with river quality models to directly assess the impact of discharges on river quality.

2.3.1 Retention and detention structures

Detention basins were one of the most popular controls adopted by NURP projects. They can be divided into wet basins and dry basins, depending on their normal state during dry weather. Dry basins were not found to be effective at pollutant removal (US EPA, 1983b), but Grizzard et al (1987) observed that with careful design removal rates could be improved. Maestri and Lord (1987) highlighted the danger of resuspension of previously settled solids, and concluded that dry basins were not effective as pollution controls.

Wet basins include ponds, lakes, and enlarged sections of drainage systems designed so that runoff from an individual storm displaces all or part of the stored water and the remainder is retained until the next event. These basins have been found to be efficient at reducing soluble and particulate pollutant loads through biological action and sedimentation (US EPA, 1983b). The observed pollutant removal efficiency was found to be very sensitive to the basin design, and a design procedure has been proposed by Maestri and Lord (1987) for efficient wet basins. The cost of construction and maintenance of detention basins is high compared with other pollution controls (Hall and Ellis, 1985), particularly as settled sediment has to be removed and disposed of safely.

Retention basins include on-line and off-line percolation ponds, infiltration systems, underground perforated pipes, and vegetated swales. They remove pollutants by sedimentation, physico-chemical and biological processes, similarly to detention basins but with an added filtering mechanism. NURP (US EPA, 1983b) found such devices generally capable of efficient pollutant removal, but attention was drawn to the dangers of contamination of groundwater in susceptible areas. Yousef and Wanielista (1986) reported removal rates in an on-line pond of 95 and 55% for dissolved and particulate lead, 88 and 96% for zinc, and 50 and 77% for copper. Nutrient removal rates were also high. Retention ponds have the same disadvantages as detention basins; that is high construction and maintenance costs, and high space requirements.

Infiltration systems, such as porous roads and car parks, vegetated swales, and perforated underground pipes have received a mixed reception in the literature. Porous road surfaces have the added advantage of improving safety by reducing skidding. Porous paving can be constructed with an impervious base so that runoff eventually enters the sewer system, but the high cost of such surfaces is restrictive and they are only constructed where spray suppression is required such as on airport runways (WRc/WAA, 1986). Hogland and Niemczynowicz (1987) conducted a study in Sweden using a 'unit superstructure' in which water from a car park passed through a permeable layer to a macadam layer containing a drainage pipe. They found that most of the chemical pollutants accumulated in a geotextile layer at the base of the structure, thus improving runoff quality. Reduced pollutant loadings have been observed in a similar UK study (Pratt et al, 1989). Both studies have indicated, however, that the infiltration capacity of such surfaces may be reduced over long periods, as clogging occurs. Shaver (1986) also commented on clogging and sealing of infiltration structures, and suggested stringent sediment control during construction and operation to allow for proper performance.

Vegetated swales are wide, flat channels which transport runoff and allow infiltration. The vegetation reduces erosion, and reduces flow by increasing channel roughness. Sediment in the runoff is trapped in the vegetation (Lord, 1986), and the runoff quality is improved. Grassed swales were the control measure chosen by Maestri and Lord (1987) as the most effective for highway runoff pollution management, because of their low cost and flexibility.

2.3.2 Street sweeping and gully pot cleaning

One of the most popular control measures is street sweeping, and the effectiveness of this practice has been widely investigated (Sartor and Boyd, 1972, Field and Lager, 1975; Ellis, 1979; Sartor and Gaboury, 1984; Heaney, 1986). There is general agreement that while sweeping has aesthetic benefits, it is inefficient at removing the fine solids with which most of the pollutants are associated.

Sartor and Ellis both estimate the general pollutant removal from runoff efficiency of a well-run street sweeping program at approximately 20%, with best results achieved by the use of mechanical sweepers. Sartor and Boyd have observed that a second run along a street by the sweeper will remove approximately 50% of the remaining solids. Sartor and Gaboury state the importance of assessing street sweeping effectiveness in terms of end-of-pipe pollutant reduction, rather than in terms of quantity of material removed.

Sweeping has also been observed to redistribute solids across road surfaces. Before sweeping, the bulk of the pollutants are close to the kerb, but after sweeping they are spread over a wider area, making cleaning by a second sweeper pass more difficult. The action of the sweeper can also cause break up of large particles and increase the amount of material available for washoff by low intensity storms; thus increasing the pollutant load to the receiving water. Despite its shortcomings, street sweeping continues to be one of the most popular urban runoff pollution management practices.

Hall and Ellis (1985) suggest that cleaning of gully pots would be more effective than street sweeping, and that public awareness of problems caused by the use of fertilisers, oils and detergents could be increased by advertising campaigns. Gully pots are currently only cleaned on average once a year, irrespective of season or other factors, with negligible effect on receiving water quality. Regular and frequent gully pot cleaning would control polluting discharges far more effectively.

Maestri and Lord (1987) and Hall and Ellis (1985) have both advocated the gradual removal of the conventional kerb-grating-gully system, from residential areas and highways respectively, and their replacement with verge soakaways and grassed swales. Current opinion generally advocates the use of such low-cost measures, and an integrated approach to management of runoff quantity and quality.

It is generally accepted that the pollutants observed in urban runoff are damaging to water quality and that control measures are therefore necessary to protect receiving streams. Studies undertaken to assess the impact of runoff discharges on water quality are dicussed in section 2.4.

2.4 EFFECTS ON RECEIVING WATERS

Although runoff quality and control have received extensive coverage in the literature, very little attention has been paid to the effects of surface water discharges on stream water quality.

Early studies of runoff quality, such as those described by Wilkinson (1956) and by Weibel et al (1964), made no attempt to assess the impact of runoff flows on stream quality. Rather, runoff was considered to be a problem because pollutant concentrations and loads were comparable to domestic sewage effluent. The need for more direct assessment of the impact of surface water discharges is now recognised, principally because the high costs of pollution control cannot be justified unless undesirable effects can be demonstrated (Heaney, 1986).

Assessment of discharge impacts requires a rational framework of river classification, water quality standards, and assessment techniques. European standards exist for the protection of drinking waters, bathing waters, and freshwater fisheries (Council of the European Communities, 1975; 1976 and 1978); but many urban streams which receive storm discharges are not designated under these directives and the standards do not yet apply. New standards for the UK, based on toxicity to fish, are currently being developed to provide quality guidelines so that the effects of intermittent discharges can be overcome (Whitelaw and Solbe, 1989). This work is at present aimed at reducing pollution from combined sewer overflows and agricultural discharges.

Urban runoff discharges to streams can have physical, chemical and biological impacts, and their intermittent and unpredictable nature makes them difficult to assess. Biological monitoring has long been recognised as valuable in such cases, as stream flora and fauna respond to all the stresses placed on them by intermittent discharges. Despite this, there has been very little ecological research related to urban runoff. This was highlighted in a recent analysis of the content of papers presented at major conferences (Gujer and Krejci, 1987), which noted that even papers presented under 'receiving water effects' headings did not address the problem from an ecological standpoint. Gujer's final recommendation was that a multidisciplinary approach, involving ecologists as well as engineers, should be adopted in runoff pollution management.

The majority of published research into receiving water impacts has adopted a probabilistic approach. This is particularly true of work carried out in the US by the EPA (1984), which is discussed in detail in section 2.4.1. Other studies, in which receiving water impacts have been directly assessed, are described in section 2.4.2.

2.4.1 Probabilistic methods for analysing water quality effects

The methods and results discussed in this section are documented in the NURP final report (US EPA, 1983b) and a separate EPA report which gives a more detailed account of the probabilistic methodology used (US EPA, 1984).

<u>NURP issues</u>

The major issues addressed by NURP were

> The quality characteristics of urban runoff, and similarities or differences at different urban locations
>
> The extent to which urban runoff is a significant contributor to water quality problems
>
> The performance characteristics and the overall effectiveness and utility of management practices for the control of pollutant loads from urban runoff.

The first and third issues were considered in sections 2.1 to 2.3, and it is the second which is discussed here.

NURP recognised the need to avoid the following 'logical' sequence which had previously been applied to receiving waters:

Pollutants cause water quality problems;
there are pollutants in urban runoff, therefore
urban runoff causes water quality problems.

A three-level definition of a water quality problem was adopted, as causing one or more of the following:

Impairment or denial of beneficial uses

Water quality criterion violation

Local public perception

Public perception was not measured as it was impractical to quantify. The beneficial use selected to be of primary importance was the protection of aquatic life. Some of the individual NURP projects measured stream quality during storm events downstream of outfalls, and a few examined the effects of discharges on aquatic life (see section 2.4.2); but their site-specific nature made it difficult to comment on their general relevance. Rather, a screening level analysis was developed and the individual project results were used to verify the computed concentrations and their effects.

NURP considered three types of water quality effect: rapid, short-term changes in water quality during and soon after storm events; long-term changes caused by settlement of solids or long retention times in water systems such as lakes; and physical effects, such as scour and silting. The short-term effects, including depressed dissolved oxygen concentrations and increases in toxic contaminants, were believed to be the most important. Although it was recognised that very short exposures (in minutes) to toxic pollutants could be damaging to aquatic life, it was argued that longer exposures (in hours) were more likely to be harmful. The time scale chosen for analysis of receiving water impacts was therefore the rainfall event time scale, and the average concentration of pollutants in urban runoff produced by the event was represented by the event mean concentration (EMC).

The NURP screening analysis method

The probabilistic approach adopted for analysing effects involved the use of a simple model for estimating receiving water pollutant concentrations (Fig. 2.3). Given the probability distributions of the model inputs, the probability distributions of the resulting downstream concentrations were calculated. The computed probability distributions were then used to specify the frequency with which beneficial uses were impaired, based on water quality standards and estimated effects levels for adverse use impacts.

The probability model used was that the model inputs of stream flow rate and concentration (Q_S and C_S) and runoff flow and concentration (Q_R and C_R) were stochastic random variables, lognormally distributed, which determined downstream concentrations (C_D) according to the following equation:

$$C_D = \frac{Q_S}{Q_S + Q_R} C_S + \frac{Q_R}{Q_S + Q_R} C_R$$

Use of the model permitted frequencies at which pollutant concentrations were exceeded to be determined. In order to translate these concentrations into frequencies at which beneficial uses were impaired, wet-weather water quality criteria needed to be developed. Existing criteria in the US (as in the UK) are most relevant to continuous exposure, and are not suited to intermittent discharge impact assessment. NURP therefore developed estimates for concentration levels which impaired beneficial uses when exposure was intermittent and at intervals and durations typical of runoff events (Table 2.6). These estimated effect levels were quoted in terms of 'threshold' and

Fig. 2.3 Schematic representation of urban runoff discharges entering receiving stream (US EPA, 1984) 'significant mortality' concentrations, where threshold levels were those estimated to cause mortality of the most sensitive individual of the most sensitive species; and significant mortality levels were ranges reflecting mortality of 50% of the most sensitive species and mortality of the most sensitive individual of the 25th percentile sensitive species.

TABLE 2.6. ESTIMATED EFFECT LEVELS FOR INTERMITTENT EXPOSURE USED IN NURP SCREENING ANALYSIS

Contaminant	Water Hardness mg/l $CaCO_3$	EPA max	Estimated effect level (ug/l) Threshold	Significant mortality
Copper	50	12	20	50 - 90
	100	22	35	90 - 150
	200	42	80	120 - 350
	300	62	115	265 - 500
Zinc	50	180	380	870 - 3200
	100	321	680	1550 - 4500
	200	520	1200	2750 - 8000
	300	800	1700	3850 - 11000
Lead	50	74	150	350 - 3200
	100	172	360	820 - 7500
	200	400	850	1950 - 17850
	300	660	1400	3100 - 29000

Although the analysis method permitted computation of instream pollutant concentrations which incorporated the effects of upstream concentrations, lack of data dictated that upstream concentrations should be set to zero. The analysis was based on eight regions of the US, with typical rainfall and stream parameters assigned to each. The quality characteristics of the runoff were taken from analysis of the NURP data set, using low (10th percentile), average (50th percentile), and high (90th percentile) range site EMCs. This was considered necessary because of an observed tendency of some sites to have EMC values at the top or bottom of the range.

Results of the NURP screening analysis

The results were presented as comparison plots showing the recurrence interval at which specified effects levels were exceeded. An example is given in Fig. 2.4. for copper. Each bar represents a geographical region, and for each of these the results are presented for two drainage area ratios. At computed recurrence intervals of 10 years or more the results were not considered reliable and all the plots were terminated at 50 years.

The following example interpretation is from the middle plot of Fig. 2.4, for area 1 and a DAR of 10:

> The EPA maximum level is exceeded once every 0.02 years, or 50 times a year.
>
> Threshold concentration levels are reached once every 0.05 years, or 20 times a year.
>
> Significant mortality levels are exceeded twice a year for the less severe effect and about once every 5.5 years for the more serious impact.

Inspection of all the results presented in this way made it clear that it was unrealistic to make a general statement on whether or not urban runoff constituted a problem in rivers and streams. Impacts varied widely, and could therefore only really be considered on a site-by-site basis. The following generalisations were made concerning impacts:

a) Water hardness is possibly the most important factor in assessment of effects on aquatic life.

b) Drainage area ratios are also important as they reflect the magnitude of stream flow.

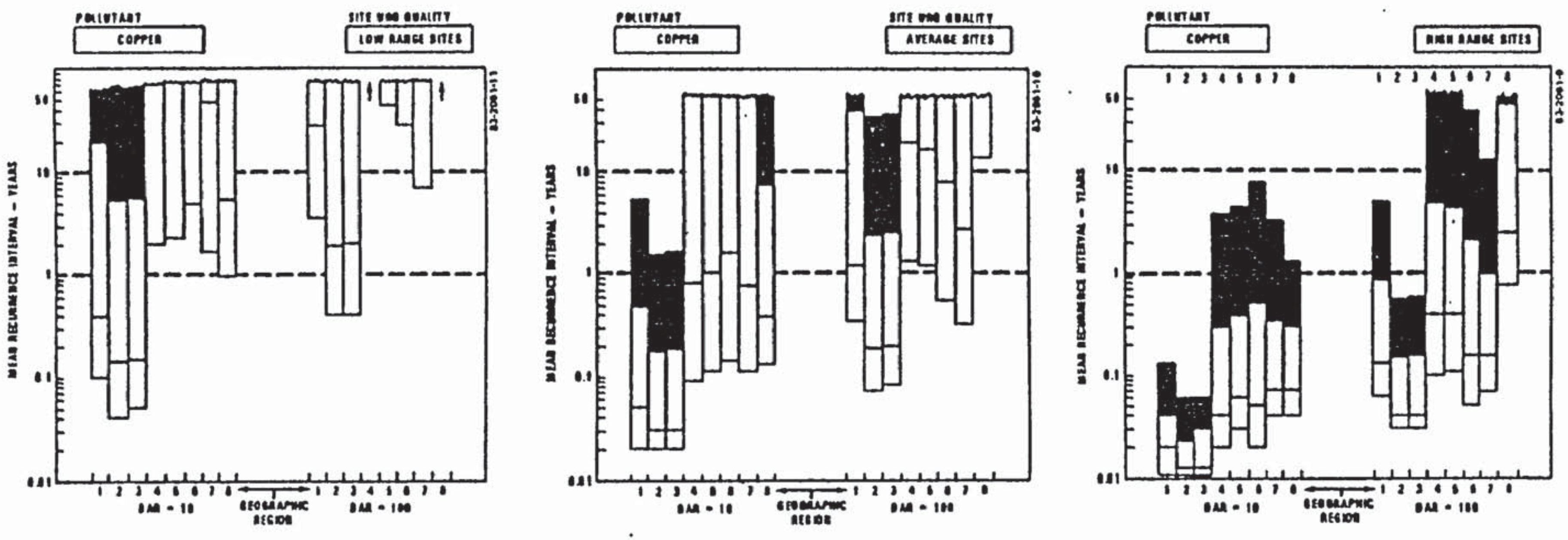

Fig. 2.4 Exceedance Frequency for Stream Target Concentrations of Copper (after US EPA, 1983b)

c) The quality characteristics of the site (whether low, average or high range) have a significant influence on impact.

d) Threshold effects levels in streams are reached more than once a year on average.

e) 50% mortality of the most sensitive species occurs more than once every 10 years on average.

f) In reality background stream concentrations are greater than zero and could result in more severe conditions than those identified.

Using these general observation, copper was tentatively identified as the key toxic pollutant in runoff. Lead and zinc did not cause a problem under any conditions in which copper did not, and copper is a more generic toxin than either lead or zinc. Copper was estimated to be 50% in solution in runoff, as was zinc, with lead approximately 10% in solution. Control measures based on reduction of copper concentrations should therefore also be effective for zinc and lead. Finally, NURP noted that long-term effects not considered in the screening analysis might be important for particulate pollutants such as lead, but no comment on their significance was made.

The methodology described above was tested using measured in-stream concentrations to confirm that the computations were valid, with good results. The validity of the estimated effect levels was also tested, using the limited data collected by individual NURP projects. Although presented tentatively, the available biological data and predicted effects did not disagree.

NURP conclusions

The conclusions reached by NURP on receiving water quality effects were based on the screening analysis described above and the observations and conclusions of individual projects. In general NURP concluded that effects were highly site-specific and dependent on the type, size and hydrology of the water body; the runoff quantity and quality characteristics; the designated water use; and the pollutant concentration levels estimated to affect that use. As only rivers and streams have been considered here, only the relevant NURP conclusions are listed below.

1. Frequent exceedences of heavy metals quality criteria for freshwater aquatic life are produced by urban runoff.

2. Although a significant number of problem situations could result from heavy metals in urban runoff, levels of freshwater aquatic life use impairment suggested by the magnitude and frequency of criteria exceedences were not observed.

3. Copper, lead and zinc appear to pose a significant threat to aquatic life uses in some areas of the US. Copper is suggested to be the most significant of the three.

4. Organic priority pollutants in urban runoff do not appear to pose a general threat to freshwater aquatic life.

5. The physical aspects of urban runoff, e.g. erosion and scour, can be a significant cause of habitat disruption and can affect the type of fishery present.

6. Several projects identified possible problems in the sediments because of the buildup of priority pollutants contributed wholly or in part by urban runoff.

7. Coliform bacteria are present at high levels in urban runoff and can be expected to exceed EPA criteria during and immediately after storm events in most rivers and streams.

8. Domestic water supply systems with intakes located on streams in close proximity to urban runoff discharges are encouraged to check for priority pollutants which have been detected in urban runoff, particularly those in the organic category.

The utility of the NURP approach and its applicability to the UK

The probabilistic methodology presented by NURP provides a useful planning tool for evaluating the effects of urban runoff discharges and a framework from within which to assess discharge control options.

NURP has been criticised for not evaluating the water quality significance of discharged contaminants, particularly for not making more extensive use of bioassay techniques and for not giving more consideration to water quality standards (Lee and Jones, 1981). These criticisms have been answered by Myers et al (1982), who replied that NURP had addressed all the relevant fundamental issues, and agreed that the setting of standards required further attention. Lee and Jones neglected to criticise directly a further issue which has been acknowledged by the EPA as one which merits further study (Field and Turkeltaub, 1981); that of the long-term influence of sediment-associated pollutants. Studies have shown that pollutants can build up in sediments in receiving waters, where they can exert prolonged oxygen demands and form a reservoir of toxic metals which can be released into solution. Long-term effects of sediments were not included in the NURP analysis.

Similar probabilistic techniques to those developed by the EPA have been suggested for use in the UK (e.g. Aspinwall and Ellis, 1986), but have not been adopted as standard procedures and at present there is no consensus in the industry on the basic working parameters for evaluating river impacts. An interim procedure for estimating pollutant loads from combined sewer overflows is described in the second edition of the Sewerage Rehabilitation Manual (WRc/WAA, 1986); and can be used in conjunction with simple water quality planning procedures to calculate acceptable loads to rivers. This procedure is currently being extended to include more sophisticated deterministic sewer and river quality models, which will eventually be used with water quality standards designed to take account of intermittent storm discharges. Meanwhile, probabilistic tools for assessing pollution loads from surface water systems cannot be used until UK urban runoff quality characteristics are better known; and in the absence of a major data collection project small, site-specific studies provide the only means of assessing the impact of urban runoff discharges. Studies of this type in the UK and elsewhere are described in section 2.4.2.

2.4.2 Chemical and ecological studies of receiving water impacts

As stated earlier, there are very few published accounts of receiving water impact assessment using direct measurement of chemical or ecological parameters. Those which have been carried out are generally restricted to a single site, and with the exception of the NURP projects they have not been co-ordinated and have used widely differing methods.

Individual NURP projects

Several individual NURP projects studied the effect of urban runoff on aquatic life in streams. The project at Bellevue, Washington, concluded that any adverse effects were due to bed scour and deposition rather than chemical toxicity. It was also reported, however that heavy metals accumulating in sediments caused problems for fish. The same observation was made by the Denver project, where concentrations of lead, copper, zinc and cadmium exceeded water quality standards throughout the monitored storms.

The Tampa NURP project included extensive bioassay tests, but no acute or chronic toxicity to test organisms was revealed. The Rapid City project tested benthic organisms and fish in the stream receiving runoff from the entire city, but observed no adverse effects. The study reported a healthy fishery in the stream, which was unaffected by increasing inputs of runoff as more of the city contributed to the flow.

In San Jose, California, the chemical and biological changes to Coyote Creek were monitored. Upstream of the urban area, the river supported a variety of fish and benthic macroinvertebrates; but communities in the urbanised portion were restricted to pollution-tolerant algae and worms. The sediments in the urban reaches contained 10 times as much lead and 9 times as much arsenic as those taken in non-urban reaches. BODs, orthophosphates and hydrocarbons were also found in higher concentrations. The study also looked at metals concentrations in algae and other organisms, and again found higher levels in urban areas than in non-urban ones. Lead and zinc concentrations in some organisms were 500 times greater than in the water.

UK studies

Studies which have measured the effects of urban runoff on UK streams have been carried out by Extence (1978); Dussart (1984); Shutes (1984); Davis and George (1987) and Bascombe et al (1988).

Many studies have been published in which the effects of suspended solids, organic enrichment, or metals pollution on stream biota have been assessed, some of which have recently been reviewed by Hellawell (1986) and Mance (1987); but they do not directly address the issue of urban runoff. Methods for evaluation of the toxicity of specific pollutants to fish are well documented, and provide the basis for relevant water quality standards (e.g Alabaster and Lloyd, 1982); but these standards are based on results of long-term exposure to pollutants rather than the short exposure times typical of streams receiving urban runoff. In addition, the applicability of toxicity tests to organisms other than fish has not been evaluated. Field observation of the ecological effects of urban runoff discharges therefore remains a simple and direct method of impact assessment.

Extence (1978) studied the effects of runoff from the M11 motorway in Essex during the period of its construction. The runoff from the construction site was carried a short distance by a brook before discharging into the River Roding. Extence measured the concentrations of contaminants and took samples of macroinvertebrates in the River Roding upstream and downstream of the confluence; and measured contaminant levels in the brook. Mean suspended solids and iron levels were, respectively, approximately three times and twice those in the river upstream of the confluence; resulting in elevated concentrations in the river downstream. The input of solids was also apparent in the substratum downstream of the confluence, which was originally stony, but during the period of study became covered in sand and silt. No differences were found in pH, BOD, nitrate, or phosphate concentrations. The suspended solids input and the resulting habitat disruption were thought to be the cause of observed changes in the river ecology. Caseless caddis, leeches, molluscs and the mayfly *Caenis* were markedly less abundant downstream of the confluence, even though some species present upstream are generally regarded as silt-tolerant. The alga *Cladophora* disappeared from the river downstream of the confluence during the study. Extence concluded that discharges associated with the motorway construction had undesirable effects on the quality of the receiving stream, and that erosion control at such sites should be introduced.

Dussart (1984) reported on a survey carried out by North West

Water, following a complaint by an angling club that trout stocks were being adversely affected by motorway runoff. The aim of the survey was to determine whether there was a measurable impact of motorway runoff on stream biota. 7 sampling sites were chosen from small streams in Cumbria and Lancashire which were crossed by the M6 and had similar characteristics upstream and downstream of the motorway. Samples were taken in winter, so that the effects of road-salting operations could be considered, and at each site rocks were sampled for later identification and counting of algae. Analyses of variance showed that at sites downstream of the motorway there were significant increases in the number of algal species, algal abundance, and species diversity. Since the streams studied were nutrient-poor, it was suggested that the runoff from the M6 was a source of nutrients for the plant communities downstream; in which case similar increases in diversity and abundance might be expected at higher trophic levels. Macroinvertebrates, however, showed a decrease in the number of species downstream of the motorway. Dussart concluded that the motorway runoff had a significant impact on the stream quality, but as the mechanisms causing the impact were beyond the scope of the project, they were not investigated further.

Shutes (1984) studied benthic macroinvertebrates in the Silk Stream, a tributary of the River Brent in London. The Silk Stream and its two tributaries - Dean's Brook and Edgware Brook - drained an area of over 5000 ha, 65% of which was urban, including two major roads, part of the M1 motorway, and the Grahams Park Estate previously monitored by Ellis (section 2.1.1). 6 sampling stations with riffles were selected. Station 1 was above the M1 outfall, and the remaining stations were numbered sequentially so that station 6 was the furthest downstream. Benthic macroinvertebrates were sampled monthly for a year. Water samples were taken at the same time, and pH, temperature, dissolved oxygen and conductivity readings were made on site. Analysis of the water samples showed the stream to be generally clean, and a typical chalk stream, with very high hardness levels. Water samples were not taken in wet weather, however, so it was not possible to directly relate observed changes in macroinvertebrate communities to storm discharges. Crustacea were present at the upstream stations, but numbers fell with distance downstream, and *Gammarus* were absent from the lower stations. Stoneflies and mayflies were absent from all stations, and caddis were found only at station 1. Numbers of molluscs also fell from stations 1 to 6, with the exception of *Lymnaea peregra*, which was believed to have migrated upstream from the nutrient-rich receiving waters of the Welsh Harp lake. The data were used to calculate the Trent Biotic Index, Chandler's Score, and a Community Diversity Index. All three indices confirmed that the stream became progressively more polluted downstream. Finally, a predictive model based on physico-chemical parameters was used to estimate the Diversity Index at each site. The model predictions were close to the observed values at all except stations 5 and 6, where diversity was much lower than expected given the physico-chemical characteristics of the stream. The runoff from Grahams Park entering the stream was offered as an explanation for this result. Shutes's results, although not direct measurements of runoff effects, indicate strongly that runoff is responsible for at least some of the observed deterioration in water quality of the Silk Stream.

Davis and George (1987) also used benthic macroinvertebrates to assess changes in quality in the River Roding, following the earlier work of Extence described above. Water and macroinvertebrate samples were taken monthly, and as well as identification and counting of animals, macroinvertebrate and plant tissue was analysed for metals. Water samples were not taken in wet weather. Davis and George reported statistically significant differences in quality between sites and related these to sewage effluent and runoff discharges, but the changes were very small and although the continuous discharge of sewage effluent could have a measurable effect on quality the observed changes could also be attributed to normal sampling and analytical error. The macroinvertebrate data was used to calculate Chandler's Score and Simpson's Diversity Index. Chandler's Score, based on organisms' tolerance of organic pollution, decreased markedly downstream, and as expected it correlated well with dissolved oxygen measurements. The Diversity Index followed a similar pattern. Measurement of metals levels in macroinvertebrate and plant tissue revealed more subtle differences between sites than the water analyses, with copper, lead and zinc concentrations highest in plant and animal tissue at sites most affected by urban and motorway runoff. Davis and George concluded that the changes in macroinvertebrate community structure could be largely explained in terms of organic enrichment by sewage effluent, and demonstrated by the use of biotic indices. Such methods were not suitable for detecting urban and motorway runoff effects, which were better evaluated by analysis of plant and animal tissue for metals.

Bascombe et al (1988) also measured metals levels in macroinvertebrate tissue. Caged populations of *Gammarus pulex*, *Asellus aquaticus* and *Lymnaea peregra* were placed at 7 sampling sites along the Salmon's Brook, London, including clean 'background' sites and sites immediately downstream of major surface water outfalls. The cages were intended to reduce the impact of habitat disturbance caused by intermittent discharges, and provide a stable physical environment for monitoring chemical effects. Levels of copper, zinc and lead in tissue were found to increase over 5 week monitoring periods at the sites downstream of the outfalls; with little change at the background site. The largest increases were observed downstream of an industrial area and a storm sewage overflow. Bascombe concluded that the use of caged species was a more reliable method of assessing toxic effects than monitoring of organisms in the natural environment, and that it allowed better assessment of metal contributions from dissolved and particulate sources because the organisms were prevented from migrating. On the basis of the reported bioaccumulation of metals in tissues, the runoff entering the brook appears to have an undesirable impact. The full impact of discharges can not be assessed by a study such as this, however, as physical disturbance is not taken into account.

Other studies

Similar studies to those carried out in the UK have been described by Barton (1977), Taylor and Roff (1986) and Lord (1987) in the US and Canada; and by Roos (1986) in the Netherlands.

Barton (1977) investigated water chemistry, sedimentation, fish populations, and macroinvertebrates in a small stream in Ontario during construction of a major highway. Suspended solids levels rose to a maximum of 1390 mg/l during construction, and sedimentation increased downstream of the construction site. The settled sediments were largely sand from the construction site itself, and both suspended solids and sedimentation decreased downstream of a series of ponds just below the construction site. No changes in water chemistry were recorded during the study. The standing crop of fish was reduced just below the construction site, but not further downstream, and Barton attributed this to migration out of the area of high suspended solids, as there was no evidence of fish mortality. Both suspended solids and fish populations returned to pre-construction levels when the highway was completed. The overall numbers and diversity of macroinvertebrates were not affected by the construction work, but there was a change in the species composition of the community. Mayflies and net-spinning caddis were reduced in numbers and replaced by increased water mite and case-building caddis populations. Barton attributed this to increased drift caused by high suspended solids upstream, as the ponds prevented siltation downstream.

Barton's work was continued by Taylor and Roff (1986) who studied the same stream for long-term changes in ecology. They found that changes in macroinvertebrate numbers and diversity did occur, but the effects were delayed and not apparent until soil cover had been re-established following construction work. Taylor and Roff observed large, apparently permanent increases in organism numbers and suggested that altered drainage patterns had resulted in an increased supply of nutrients to the stream. Other changes which occurred slowly were the return of silt-intolerant species such as some mayflies, as the construction silt was gradually washed out of the stream. Numbers of fish also continued to increase following Barton's work, and exceeded pre-construction levels. Taylor and Roff concluded that the long-term effects of highway construction were complex and relatively permanent. Barton's observation that the stream had largely returned to its original state following the cessation of construction work was modified in the light of this long-term investigation, and the permanent effects appear to be part due to runoff and altered drainage patters.

Lord (1987) described the US Federal Highway Administration (FHWA) research program to protect the environment from highway runoff. The program had four phases with the following objectives:

1. Identify and quantify the constituents of highway runoff.

2. Identify the sources of these pollutants and migration paths from the highway to the receiving water.

3. Analyse the effects of these pollutants in receiving waters.

4. Develop the necessary abatement/treatment methodology for objectionable constituents.

Phase 3 of the FHWA project had specific objectives which included assessment of the effects of highway runoff on stream biota, and provision of guidelines for field studies to assess such effects. Field and laboratory studies were carried out at two stream sites, and it was found that highway runoff from low to medium traffic areas there was minimal impact. Much of the work was carried out under laboratory conditions, using undiluted runoff and testing its toxicity to organisms over several days. Effects other than toxic ones were therefore not measured, and neither were delayed effects of accumulated toxic substances in sediments. *Gammarus* and some mayfly nymphs were found to be sensitive to exposure to the undiluted runoff in the laboratory.

Roos (1986) reported a study in Lelystad, The Netherlands, in which the biological effects of surface water discharges to canals were assessed. Because of strict controls during sewer construction, the system had very few connections to the foul sewers and this, coupled with a high standard of street cleanliness, made it a good site for assessing minimum biological effects. Samples of macroinvertebrates were taken at two-weekly intervals from May to September 1983 from 12 sites in two study areas. Using saprobic and diversity indices, Roos detected limited but distinct differences in species composition in the vicinity of outfalls; with indicators of moderate organic pollution more abundant at outfalls and indicators of mild organic pollution rarely found. Roos concluded that the water in the vicinity of the surface water outfalls was organically enriched, despite the relative cleanliness of the catchment and low incidence of connections to the foul system.

2.5 RESEARCH NEEDS

Several papers have been published in the last few years which have summarised current research in urban storm drainage and identified future research needs. Five recent examples are briefly discussed here, with reference where appropriate to research projects which subsequently fulfilled identified needs.

Following the Second International Conference on Urban Storm Drainage in 1981, a Joint Committee on Urban Storm Drainage was formed under the auspices of IAWPRC and IAHR. The Committee's technical report on the conference (Colyer and Yen, 1983), as well as providing an overview of current practices, summarised future trends and needs. One of the major identified trends was increasing interest in pollution control aspects of drainage systems, and some of the specific topics relating to this were listed as follows:

Collection of storm water quantity and quality data with well planned programmes.

Analyses of the data for characterisation and generalisation and for evaluation of mathematical models.

Identification of the representative water quality parameters in urban storm water pollution and characterisation and normalisation of these parameters from local data for general use.

Investigation of the need for and significance of modelling the biological and chemical processes in urban storm water transport.

Investigation of the performance and preferability of combined or separate storm sewer systems under different conditions.

Colyer and Yen also called for funding and support of technology development from research to implementation, but recognised the difficulties in obtaining such support. Many of these topics have since been addressed by NURP in the US, but nothing on this scale has been attempted in the UK, particularly in the field of data collection.

Following the publication of the NURP results, Heaney (1986) projected research needs in the US for the following five years and grouped them under four headings: urban runoff characterisation, water quality effects, control effectiveness, and decision-making models. Requirements listed under characterisation of runoff quality were largely concerned with further analysis of the NURP database rather than collection of more data. Control options and decision-making models concerned with pollution from runoff can only sensibly be developed once undesirable effects have been demonstrated, leaving water quality effects and the development of wet-weather quality standards as the most urgent research need. UK research needs are similar, but must include data collection to characterise runoff quality if adverse effects are detected.

At a recent workshop on urban runoff pollution, a session was held specifically to identify future research needs (Torno, 1986). The following three items were the most commonly listed when the participants were asked to identify what they considered to be the four or five most important research needs related to urban runoff pollution:

> The understanding of the processes and mechanisms, including the related mathematical models, of pollutant accumulation and transport in urban runoff.
>
> The effects of metals and other toxic pollutants on aquatic biota, including the means to monitor these effects.
>
> The impacts of urban runoff on receiving waters, including the impairment of beneficial uses. This includes the modelling of these aspects.

Again, the emphasis is on better understanding of pollution and how it affects receiving waters. Implicit in such an approach, although not specified, is the development of appropriate water quality standards.

Similar conclusions were reached by a workshop on future activities needs held at the end of a recent conference in the US (Barnwell and Jones, 1986). The Workgroup on Pollution Sources and Receiving Water Impacts identified several areas in which research was needed, all relating to evaluation of storm runoff impacts. The common theme throughout the Workgroup report was the need for development and standardisation of methods used to evaluate impacts, together with a better understanding of the polluting processes involved and better distribution of data and information on all aspects of runoff management.

Both workshops described above were held in the US, and although UK delegates participated, this reflects the higher level of activity and funding in the US relative to the UK. The research needs identified by the workshops reflect the state-of-the-art in the US rather than the UK, which, largely as a result of NURP, is more advanced. Despite this, the need to evaluate impacts of urban runoff on stream quality is not restricted to the US, and the same requirements for development of wet weather standards and development and standardisation of methods exist in the UK and Europe. At the Fourth International Conference on Urban Storm Drainage, held in 1987, Gujer and Krejci identified areas where more research is needed into ecological aspects. Their major recommendation was for a more integrated, multidisciplinary approach to urban drainage; involving ecologists as well as engineers, consideration of pollution from its source to its manifestation, and problem-solving on a small scale so that appropriate solutions were found.

CHAPTER 3

PROJECT FORMULATION AND SPECIFICATION

The literature review described in the previous chapter confirmed the need for research into the effects of urban runoff discharges on receiving stream quality. Biological surveillance had been identified as a valuable means of water quality assessment where the effects of intermittent storm discharges were being studied.

The initial aim of the project was to determine whether discharges from surface water sewers had any measurable impact on stream quality, using commonly applied quality indices.

As it was known that many surface water systems have illegal connections to the foul system or are used to dispose of industrial wastes, it was considered unrealistic and undesirable to exclude systems where cross-connections or industrial malpractice were known to exist. This was reflected in the title

> Assessment of the impact of discharges from surface water sewers on receiving water quality

A broad based approach using chemical and biological surveillance was chosen to assess discharge impacts, by upstream-downstream studies at as many outfall sites as possible. This broad approach was deliberately chosen to contrast with other UK studies in which a small number of sites had been investigated in detail. A target of investigating 50 sites was set. For practical purposes, so that this number of sites could be studied, it was intended that sampling and laboratory work should be kept as simple as possible.

No attempt was to be made to investigate the processes which produced any observed impact. For this reason it was not necessary to monitor quantity or quality of discharges from outfalls.

Upstream-downstream biological sampling was selected as the major technique to be applied. Monitoring was to be concentrated in the summer of 1987, as adverse effects on stream biota are most apparent during the summer months. This would also allow summer 1986 to be used for training; and for the evaluation and selection of appropriate techniques. Samples were to be taken at the outfall as well as upstream and downstream, so that the effects of physical disturbance could be assessed. The locations of the sampling stations would depend on the site characteristics. The indicator organisms to be sampled would probably be benthic macroinvertebrates, and their identification in the laboratory would initially be to family level only, which would be sufficient for the calculation of simple indices. Techniques for sampling and preservation of organisms, and for subsequent analysis of the data would be selected and tested in a pilot study carried out in summer 1986.

For chemical analysis of stream water to be meaningful, samples would need to be taken during storm events, and this would require the use of automatic samplers. It was decided that the scale of the project was too large for this to be practical, and that plants and sediments would be sampled; as they would reflect the stream quality during periods of discharge. The alga *Cladophora* was thought to be suitable for sampling because it is commonly found in urban streams. Metals were selected as the chemical parameter to be determined on the algae and sediment samples, because they were known to be present in urban runoff in the dissolved and particulate phases. Metals had also been cited by other studies as a likely cause of adverse effects on receiving waters.

To supplement the field work programme, catchment data collection was planned, both on site and off, so that observed impacts could be related to such simple parameters as catchment area and land use. Specific data requirements would depend on the data availability, and were to include catchment area, land use, impervious proportion, stream catchment area upstream of the outfall, and the stream quality classification. Observations made at the time of sampling would be recorded and would include a description of any discharge and a description of the substratum and plants present at the sampling stations.

The specification went through several iterations before the field work programe began, and the techniques finally selected are described in Chapters 5 and 6. Many of the techniques used were investigated in a pilot study, which is described in Chapter 4.

CHAPTER 4

PILOT STUDY

The purpose of the pilot study described in this chapter was to select techniques for use in the main body of the project, provide training in these techniques, and assess the feasibility of the proposed practical work.

Emphasis was placed on biological sampling and the treatment of biological data, as these were considered more difficult than the collection of sediment and algae samples for metals analysis.

The pilot study was carried out in two parts: initial selection of techniques, followed by a feasibility study in which the techniques were evaluated.

4.1 SELECTION OF TECHNIQUES

Techniques evaluated in this part of the pilot study were for biological sampling and the interpretation of biological data.

Benthic macroinvertebrates were selected as the indicator organisms to be studied, for the following reasons:

a) Popularity - macroinvertebrates are widely used to assess water quality and many indices exist to aid data analysis.

b) Ease of sampling - simple standard methods exist for sampling macroinvertebrates.

c) Ease of identification - good taxonomic keys are available for macroinvertebrates.

d) Wide distribution - macroinvertebrates are well distributed which facilitates site-to-site comparisons.

4.1.1 Biological sampling

Two methods for sampling benthic macroinvertebrates were considered: kick-sampling using a handnet, and the Aston cylinder sampler. Handnet sampling is widely used for qualitative work, while cylinder sampling is a quantitative method (Standing Committee of Analysts, 1980). Work carried out at Aston University had indicated that a 30 second kick sample was roughly equivalent to a 0.1 m^2 cylinder sample (Hawkes, 1986).

Two river reaches known to support a variety of macroinvertebrates were chosen as training sites: the River Cole at Kingshurst, Birmingham, and the River Blythe at Temple Balsall. At each site, 10 samples were taken; 5 with a standard handnet and kick sampling for 30 seconds, and 5 with an Aston Cylinder Sampler.

The samples were returned to the laboratory and the macroinvertebrates were identified in the fresh samples using a general taxonomic key which enabled identification of major taxa, usually to family level.

The numbers of taxa recovered by the two sampling methods used are tabulated below (Table 4.1). The total number of taxa at each site was taken as the number observed in the 10 samples taken at that site. At both locations, a higher percentage of taxa was recovered using the cylinder sampler than by hand net sampling. Overall, each hand net sample yielded a mean of 37.6% of the taxa present, compared with 61.6% for each cylinder sample.

TABLE 4.1 NUMBERS OF TAXA RECOVERED BY HAND NET AND CYLINDER SAMPLING

	HAND NET					CYLINDER SAMPLER				
	HN1	HN2	HN3	HN4	HN5	CS1	CS2	CS3	CS4	CS5
RIVER COLE										
No. of taxa in sample	5	4	4	4	6	9	5	8	6	7
% of total site taxa	42	33	33	33	50	75	42	67	50	58
mean % of total taxa			38.2					58.4		
RIVER BLYTHE										
No. of taxa in sample	11	6	9	6	5	12	7	9	14	-
% of total site taxa	55	30	45	30	25	60	35	64	100	-
mean % of total taxa			37.0					64.8		

Based on the information presented in Table 4.1, the cylinder sampler was selected for future use in preference to the hand net. Use of the cylinder sampler had the added advantage of providing samples which could be analysed quantitatively if required, whereas the hand net was essentially a qualitative tool. A full description of the cylinder sampler and its use is given in Chapter 5.

4.1.2 Interpretation of biological data

The problems of presentation and interpretation of biological survey data are well documented and have recently been summarised by Hellawell (1986).

It is common practice to interpret the raw data using a recognised index, to aid comprehension by non-biologists. Such indices fall into three categories: pollution indices, diversity indices, and comparative indices.

Pollution indices

Most pollution indices are based on the observation that groups of macroinvertebrates progressively disappear with increasing pollution load. With the exception of a few simple indices, such as the ratio of *Gammarus* to *Asellus* in assessment of the level of organic enrichment (Hawkes and Davies, 1971), pollution indices can be complex to derive.

Commonly used pollution indices include the Trent Biotic Index (Woodiwiss, 1964), Chandler's Biotic Score (Chandler, 1970), and the Biological Monitoring Working Party (BMWP) score (National Water Council, 1981). The Trent Biotic index and Chandler's Biotic Score both require identification of some groups of macroinvertebrates to species level, and Chandler's Score also requires semi-quantitative data. Without laboratory support for identification of macroinvertebrates, it was considered that the use of these indices would restrict the number of samples and therefore the number of outfall sites which could be studied.

The BMWP score (National Water Council, 1981) is a simple system based on the presence or absence of different macroinvertebrate taxa. It generally requires identification to family level only, thus reducing the time needed for sample analysis. It is a purely qualitative tool, and thus does not reflect differences in numbers of individual organisms. The scores allocated to organisms are based on the organisms' tolerance to organic pollution. Despite the BMWP score's qualitative nature and its bias towards organic pollution, it was selected for use in this project because it is simple to apply, widely used, and widely understood. Details of BMWP score calculation are given in Chapter 6.

Diversity indices

Indices of macroinvertebrate community diversity relate the number of different species present to the total population. Changes in a diversity index can be used to assess the degree of environmental stress, with the premise that communities under stress undergo a reduction in diversity.

An advantage of diversity indices is that the intensity, rather than the nature, of environmental stress is assessed. A disadvantage for this project is that they require identification to species level and counting of individuals. In addition, Hellawell (1986) has reported differing opinions of the relative merits of diversity indices, which makes selection of an index difficult.

The time required for identification and counting of macroinvertebrates was considered too great for diversity indices to be used in the project.

Comparative indices

This type of index is particularly suited to studies such as this where spatial comparisons between macroinvertebrate communities are to be made. Comparative indices compare the species composition of two communities, and may also take abundance into account. Despite the fact that they require identification to species level, it was decided that the use of comparative indices should be investigated further, as they could possibly be used with data at a higher taxonomic level.

A simple comparative index is Kothe's 'Species Deficit' (Kothe, 1962) which can be used to measure the difference in the number of species present above and below an outfall. This index, D, is calculated from

$$D = \frac{A-B}{A} \times 100$$

where A = number of species present upstream of outfall
B = number of species present below outfall

The index does not take into account abundance or the replacement of one species by another. It was decided that in view of this, a count of the number of taxa present at each station would probably be as informative as Kothe's Species Deficit.

Other quantitative comparison methods considered included those devised by Jaccard (1912), Kulezynski (1948), Sorensen (1948), and Mountford (1962). These differ from Kothe's Species Deficit in their consideration of joint presences of species, and are therefore more informative than a simple taxa count. They are all very similar, but Jaccard's coefficient is the simplest and was selected for use in the pilot study at the family level to assess its utility to the project. The coefficient is calculated from

$$J = \frac{c}{a + b - c}$$

where a = no. of taxa in community A
b = no. of taxa in community B
c = no. of taxa common to both communities

Two further comparative indices, devised by Raabe (1952) and Czekanowski (1913), take abundance of macroinvertebrates into account, but were not used as the counting of organisms was too time-consuming.

Weighted BMWP scores

So that abundance of organisms could be taken into account without counting large numbers, a semi-quantitative approach to macroinvertebrate data analysis at the family level would have been most desirable. However, no such method was found in the literature. An attempt was made to apply weighting factors to BMWP scores, using the levels of abundance recognised by Chandler (1970), as shown below.

$$\text{WEIGHTED BMWP SCORE} = \frac{\text{SUM OF (SCORES x FACTORS)}}{\text{SUM OF FACTORS}} \times \text{NO. OF TAXA}$$

NO. OF INDIVIDUALS	DESCRIPTION	WEIGHTING FACTOR
1 - 2	Present	1
3 - 10	Few	2
11 - 50	Common	3
51 - 100	Abundant	4
> 100	Very Abundant	5

In practice it was found that the weightings made little difference to the BMWP scores, and, as illustrated by the example in Table 4.2, large differences in numbers of organisms were not reflected in the final score. In the example the numbers of organisms in the downstream sample are reduced to one third of the upstream count. It was decided to look at weighted BMWP scores in more detail when data were available from the feasibility study.

TABLE 4.2 EXAMPLE USE OF WEIGHTED BMWP SCORES

	COUNT IN SAMPLE		WEIGHTING FACTOR		BMWP SCORE	SCORE x FACTOR	
	U	D	U	D		U	D
Ancylidae	3	1	2	1	6	12	6
Sphaeriidae	3	1	2	1	3	6	3
Oligochaeta	42	14	3	3	1	3	3
Glossiphonidae	3	1	2	1	3	6	3
Baetidae	9	3	2	2	4	8	8
Ephemerellidae	6	2	2	1	10	20	10
Hydropsychidae	3	1	2	1	5	10	5
Chironomidae	18	6	3	2	2	6	4
Gammaridae	240	80	5	4	6	30	24
SUM			23	16	40	101	66

BMWP SCORE = 40
WEIGHTED SCORE (UPSTREAM) = (101/23)x9 = 39.52
WEIGHTED SCORE (DOWNSTREAM) = (66/16)x9 = 37.13

4.2 FEASIBILITY STUDY

The purpose of this part of the pilot study was to assess the overall feasibility of the proposed practical work, with particular attention being paid to the time taken to complete various tasks. The biological data collected was to be used to investigate different score and index systems and assess their suitability for evaluating the impact of discharges.

4.2.1 Sites and Sampling

A list of 23 surface water sewer outfall sites in the Birmingham area was provided by Severn-Trent Water and five days were set aside for sampling as many of these as possible. All sites were identified before sampling using 1:25 000 maps, and 1:10 000 plans to pinpoint the outfalls more precisely.

At each site, three sampling stations were identified; one immediately upstream of the outfall, one immediately downstream of the outfall, and one approximately 100m downstream. In most cases it was not possible to sample further downstream than 100m without encountering a second outfall or a change in the stream characteristics which could mask the effects of the outfall.

The sampling already carried out in the training exercise had drawn attention to the need for replicate samples, as successive samples contained different taxa. Hellawell (1978) describes a graphical technique for estimating the required number of replicates, where the cumulative number of species is plotted against the number of samples. The shape of such curves is generally exponential. Insufficient data were available from the training exercise to apply this technique, so data quoted by Hellawell were used as a guide to the number of replicates required. Five samples were taken at each station, each representing $0.1m^2$ of river bed, or two cylinders. The samples were preserved in 2% formalin in river water, for subsequent sorting and identification of macroinvertebrates.

In the laboratory, macroinvertebrates were identified to family level as required for BMWP score calculation, and counted.

4.2.2 Feasibility study results

Twelve sites were visited during the five day sampling period, but because of practical difficulties only four were sampled. The sampled sites included two large, mixed land use catchments with industrial areas (sites 1 and 4); a small industrial estate (site 2) and a large residential estate with a dry weather sewage flow (site 3). At site 4, only two stations were sampled because of a second outfall further downstream. The problems encountered at other sites were as follows:

a) The time required for travel to and location of sites was greater than expected.

b) The time required for sampling was greater than expected.

c) Many sites were difficult to locate.

d) Access to sites was often difficult.

e) Several sites were on river reaches which were too deep to enter or to sample.

f) Many sites were located on culverted sections of watercourse.

g) One outfall was to a lake and therefore unsuitable for sampling.

These difficulties showed that a site selection programme, in which sites were visited and assessed before sampling, would be necessary to make best use of the summer sampling period.

Two of the samples taken at each station were sorted and the organisms identified and counted. Identification and counting of macroinvertebrates was difficult and time-consuming because they were preserved in formalin, which made the organisms difficult to detect and identify. Differences between sample pairs were slight, and inspection of the remaining three samples from each site showed that most taxa were represented in the first two samples. The remaining samples were not considered further. The results from each sample pair were combined to give a single set of results for each sampling station. These are presented in Table 4.3.

TABLE 4.3 FEASIBILITY STUDY SAMPLE RESULTS

	SITE 1			SITE 2			SITE 3			SITE 4	
	U	O	D	U	O	D	U	O	D	U	O
Lymnaeidae	8		3							3	8
Planorbidae							1				
Ancylidae										4	2
Sphaeriidae										2	1
Oligochaeta	8	12	7	62	20	17	3	12	3	17	1
Glossiphonidae										5	
Erpobdellidae	6		3	1	1	1	52	10	10		1
Asselidae	285	6	25	23	51	25	169	7	7		
Gammaridae					3	2				175	145
Baetidae					7	4				10	2
Ephemerellidae				1		1				5	1
Nemouridae										1	
Corixidae								1			
Hydropsychidae										6	3
Chironomidae	14	5	5		3	1	1	3		14	20
Simuliidae				1		2					
TOTAL ORGANISMS	321	23	43	88	85	53	226	33	20	242	184
TOTAL TAXA	5	3	5	5	6	8	5	5	3	11	10

U = upstream, O = outfall, D = downstream

These results were used to calculate BMWP scores, Jaccard's coefficient for the upstream and downstream samples, and weighted BMWP scores, as shown in Table 4.4.

TABLE 4.4 INDICES CALCULATED FOR FEASIBILITY STUDY SAMPLES

	SITE 1			SITE 2			SITE 3			SITE 4	
	U	O	D	U	O	D	U	O	D	U	O
BMWP SCORE	12	6	12	22	19	34	12	14	7	50	43
WEIGHTED BMWP SCORE	12.5	5.6	12.3	15.5	18.0	28.3	13.1	12.0	7.0	48.8	42.8
NUMBER OF TAXA	5	5	5	5	6	8	5	5	3	11	10
JACCARD'S COEFFICIENT		1.000			0.625			0.600			-

BMWP scores and number of taxa both showed differences in macroinvertebrate communities at sampling stations. Weighted BMWP scores also revealed differences in organism abundance: at site 1 the BMWP scores were the same for the upstream and downstream stations, but there were more organisms present upstream - as indicated by the weighted score. For sites 2 and 3, the weighted BMWP scores did not increase and decrease in parallel with the BMWP scores. The use of weighting factors seemed to reflect abundance when BMWP scores were similar, as planned, but the differences in weighted score were slight. Changes to the calculation of the weighted score were considered, but could not be fully assessed without more data.

Jaccard's coefficient provided a useful index of similarity, but without other information it did not indicate whether the taxa count increased or decreased. As for weighted BMWP scores, its utility could not be fully assessed with the limited data collected, and it was decided to review the use of indices when more data were available from the main sampling exercise.

4.3 PILOT STUDY CONCLUSIONS

The pilot study provided training, allowed techniques to be selected, and identified problems likely to be encountered during the main period of practical work. The main conclusions reached are listed below.

a) Samples of macroinvertebrates would be taken using an Aston Cylinder Sampler in preference to a hand net, as better recovery of taxa had been demonstrated using this method.

b) More thorough preparation was required before sites were sampled. Sites would have to be visited, access arranged where necessary, and sampling stations provisionally selected in advance of the summer sampling period.

c) Samples would be analysed fresh to avoid the difficulties and longer analysis times associated with the use of formalin.

d) Two 0.1 m^2 samples would be sufficient at each sampling station. This would reduce the sample load so that organism abundance in each sample could be assessed.

e) No single existing biotic index or score was suitable for this application. Comparative indices and BMWP scores were attractive because of their simplicity, but did not take abundance into account. Weighted BMWP scores were a possible solution, but required further investigation. The use of indices would be reviewed when further experience had been gained.

Details of methods employed for data collection and analysis as a result of the pilot study are given in Chapters 5 and 6 respectively.

CHAPTER 5

DATA COLLECTION METHODS

The procedures described in this chapter are for the selection of sites, the selection of sampling stations, the collection and laboratory analysis of biological samples, the collection and analysis of sediment and algae samples, the collection of field data, and the determination of catchment characteristics.

Procedures used in data analysis, including biological score calculation and statistical tests, are described in Chapter 6.

5.1 SELECTION OF SITES AND SAMPLING STATIONS

Following the difficulties encountered in the pilot study, a thorough site selection programme was planned for the project. This consisted of identification of potential sites (in consultation with local authority drainage engineers and Severn-Trent Pollution Control Officers), and visits to candidate sites to assess their suitability.

The following list of basic criteria was prepared to assist in the identification of potential sites:

a) The study catchment should be separately sewered.

b) The sewer outfall should be to a section of river with a natural bed.

c) The river should be suitable for cylinder sampling during the summer.

d) There should be no other major outfalls or combined sewer overflows close to the surface water outfall.

e) Information on the catchment and sewer system should be available.

f) The site should be within reasonable travelling distance.

Most of these criteria were assessed from maps and plans, and in many instances drainage engineers were able to assist by supplying details not recorded on plans. Candidate sites were then visited, to ensure that the criteria were met and to choose sampling stations. Visits were undertaken in winter and early spring, when banks were not overgrown and outfalls were visible.

At each site, three sampling stations were selected: one upstream of the outfall, one immediately downstream, and one approximately 100m downstream. It was not usually possible to locate the third station more than 100m downstream, because of the presence of other outfalls.

The three stations were chosen to be as similar as possible in terms of light and shade, water depth, current velocity and substratum. These steps were taken to ensure that the influence of other factors was minimised and that any observed differences could be primarily attributed to the impact of discharges from the outfall. Riffle zones were chosen wherever possible. Sites where there were major physical differences between upstream and downstream stations were rejected.

The upstream, or control station, was chosen as close to the outfall as was judged possible without the outfall affecting water quality. The outfall stations were located close to the discharge point, and in the main discharge channel so that the effects of scour and silting could be observed. The downstream stations were located where mixing of the discharge with the stream water had taken place and any physical changes caused by the outfall were absent.

Where there were two or more outfalls within about 50m of each other, they were treated as a single site and the sampling stations selected as detailed above with the outfall station at the downstream discharge point. Groups of outfalls which were more widely spaced were treated as separate sites with the downstream station of the first doubling as the upstream station of the second.

The river banks between upstream and downstream stations were examined carefully at all selected sites for outfalls not marked on sewer plans. The river was also examined upstream of the site, for at least 100m, and if any outfalls were found which could not be matched to those marked on the plans then the site was rejected.

5.2 COLLECTION AND LABORATORY ANALYSIS OF BIOLOGICAL SAMPLES

Macroinvertebrate samples were taken using an Aston cylinder sampler (Fig. 5.1), in accordance with the procedures recommended by the Standing Committee of Analysts (1982).

Cylinder samplers are open-ended cylinders which are pushed into the river bed to isolate an area from which benthic macroinvertebrates are then removed. The Aston cylinder sampler is made from stainless steel, with handles on the sides and its lower edge serrated with teeth each 1cm deep. Water flows into the cylinder through an oval aperture, covered with a mesh screen to prevent entry of drift organisms. Opposite this aperture is a second hole, with an exit port to which a sampling net is attached.

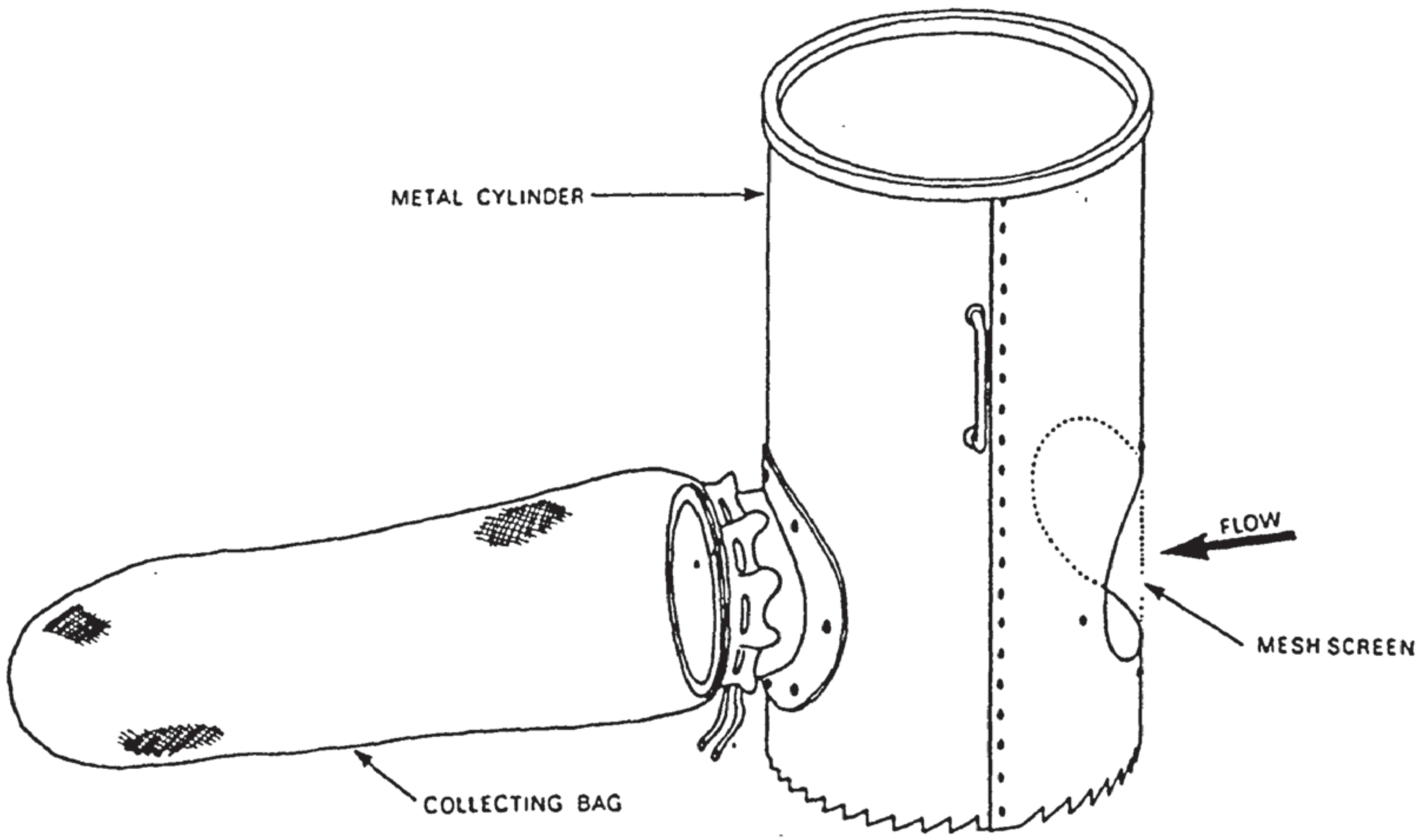

Fig. 5.1 Aston Cylinder Sampler

The sampler is placed on the river bed with the inlet screen facing upstream, and pushed into the substratum. The sampling net is extended downstream so that water flows through it. Stones in the enclosed sample area are examined and any attached animals washed into the sampling net. Small stones and fine material in the sample area are thoroughly stirred to dislodge animals into the flow and the net. The net is then detached from the cylinder and the trapped animals transferred to a container for subsequent sorting and identification.

The sampler used enclosed a sample area of $0.05m^2$. Four samples, representing a total sampled area of $0.2m^2$ were taken at each station, in different micro-habitats if appropriate. The four samples were bulked and treated as a single sample for the station. At each site or group of sites, sampling started at the station furthest downstream so that stations were not disturbed before sampling.

Animals trapped in the net were transferred to one litre plastic pails, and covered with river water. The pails were white so that the animals could be examined easily, and any fish present could be returned to the river. An initial inspection of the sample in the pail was made at the time of sampling.

All samples were analysed within three days of collection to avoid the need for preservation in formalin or alcohol, which had been found to increase analysis time in the Pilot Study. Samples waiting to be sorted were stored in a refrigerator.

Macroinvertebrates were identified in the laboratory to family level as required for BMWP score calculation (NWC, 1981), and counted.

5.3 COLLECTION AND ANALYSIS OF SEDIMENT SAMPLES

5.3.1 Sampling and initial preparation

Samples of river sediments were taken using a plastic scoop at the upstream and downstream stations of each site . At each station, the scoop was used to take samples of the top 5 to 10cm of sediment at four or more points across the channel. Where there were obvious visual differences in sediment type present at a station, care was taken to include all of them in the sample.

The sampled sediment was placed in a plastic 2mm aperture sieve over a plastic bucket, and the fine material washed through with river water. The coarse material was discarded, and the sediment in the bucket was allowed to settle. Excess water was poured off the settled sediment, and the remaining sediment and water were transferred to a plastic container. The scoop, sieve and bucket were washed with river water at each station before the sampling was undertaken.

In the laboratory, further excess water was separated from the sediment by settling and pouring off. The whole, wet sample was placed in an aluminium tray lined with greaseproof paper, and dried in a heated cabinet below 30° C, as described by the Standing Committee of Analysts (1986). After at least 48 hours, when the sample was completely dry, the dried lumps of sediment were broken up by polyethylene gloved fingers, and the dry sample thoroughly mixed. A sub-sample of at least 100g was then ground in a Tema Mill so that it passed through a 150 micron nylon mesh sieve. The dry, ground and sieved sample was stored in an airtight polyethylene container before analysis.

5.3.2 Determination of metals

Samples were prepared for analysis using the Severn-Trent method for the determination of metals in soils and sludges (STWA, 1978). This method was chosen in preference to the standard Hydrochloric-Nitric Acid digestion (Standing Committee of Analysts, 1987) which requires long digestion times and is less suitable for large numbers of samples.

Agemian and Chau (1976) reported little difference in the extracting efficiency of boiling nitric and boiling aqua regia for metals in lake sediments. The use of nitric acid alone, as in the Severn-Trent method, was considered sufficient for this application, particularly as results were not to be compared with metals figures obtained using any other method.

0.5g ground sediment was weighed and placed in a 50ml borosilicate glass tube. The sample was wetted with 1ml water and 3-4 anti-bumping granules were added. 6ml concentrated nitric acid were run into the tube slowly and any vigorous initial reaction allowed to finish. Tubes prepared in this way were placed in an electric heating block and heated until the samples were gently refluxing. The samples were allowed to reflux for 15 minutes, then cooled for 10 minutes. The samples were diluted to 25ml with distilled water (the STWA method is to dilute to 50ml, but 25ml was chosen to lower the detection limit for the metals determinations). The tubes were left to stand overnight, and the clear extract decanted off for metals analysis. Blank determinations were included with each batch of samples.

Metals were determined on the extracts by Plasma Emission Spectrosopy at Clayton Bostock Hill & Rigby Ltd., using matrix-matched standards for each of the elements measured. The analysis was covered by the laboratory's internal quality assurance scheme. The metals determined were zinc, cadmium, lead, nickel, boron, chromium, vanadium, beryllium, copper, silver, arsenic, and selenium. Results were reported as mg/kg dry sample, with detection limits of 0.5mg/kg for cadmium and 1.0mg/kg for all other elements.

5.4 COLLECTION AND ANALYSIS OF ALGAE SAMPLES

5.4.1 Sampling and initial preparation

Samples of the alga *Cladophora*, where present, were taken at the upstream and downstream stations of each site.

Composite samples, representative of the algae at each station, were obtained from plants across the channel at each location by pulling off the free-floating parts of the plants. The algae were washed in river water to remove sediment, and excess water was shaken off before placing samples in airtight plastic containers.

In the laboratory, any further separated water was removed to waste and the algae placed in aluminium trays lined with greaseproof paper. The samples were dried at approximately 30° C for at least 48 hours. The entire sample was then ground in a coffee grinder for two minutes, and returned to the drying cabinet so that no moisture was absorbed before weighing for analysis.

5.4.2 Determination of metals

Samples were prepared for analysis by the Severn-Trent method used for sediments (section 5.3.2) and metals were determined by plasma emission spectroscopy at Clayton Bostock Hill and Rigby Ltd.

The organic nature of the material being tested resulted in vigorous foaming on the addition of acid, and a silicon anti-foaming agent was added to the reaction tubes to ensure that no sample was lost. Solid material did not settle out of the extracts easily, and extracts were filtered for analysis after standing overnight.

5.5 COLLECTION OF FIELD DATA

Observations made at the time of sampling were recorded on Site Checksheets (Fig. 5.2), each of which was headed by the site reference number, the date and the time sampling commenced. The reason why items were recorded and the approach adopted in each case are described below. Photographs were taken to supplement recorded observations, and for each photograph a reference and description were included on the appropriate checksheet.

The outfall pipe diameter was measured as a check against sewer records and as an indication of the size of catchment served. The measured diameter was matched with the diameter recorded on the plans to ensure that the outfall had been located correctly. This item was not used in subsequent data analysis.

The weather (wet or dry) and whether or not the outfall was discharging were then recorded. Discharges, where observed, were characterised by appearance and odour for subsequent classification. Any evidence of past pollution, from the outfall or any other source, was also noted.

The physical effects of the discharge at each site were recorded as the visible extents of silting and scour. In practice this was often difficult to assess and these data were not used in subsequent analysis of results.

The channel characteristics width, depth, and bank slope were measured or estimated. Channel width and depth were intended to characterise the size of the receiving streams. Bank slope was

SITE REF ________ Date / /8

Time

Outfall diameter (cm) ________

Weather (Wet/Dry) ________

Outfall (Wet/Dry) ________

Discharge appearance ________________________

Discharge odour ________________________

Gross pollution/Dumping ________________________

Extent of silting (m d/s from outfall) ________

Extent of scour (m d/s from outfall) ________

Channel width (m) ________

Channel depth (m) ________

Bank slope ________

	Station U	Station O	Station D
m from outfall	________	________	________
Light/Shade	________	________	________
Macroplants	________	________	________
	________	________	________
Substratum (R/P/S%)	/ /	/ /	/ /
Ammonia (mg/l)	________	________	________

PhotoRef	Subject
________	________________________
________	________________________
________	________________________

Fig. 5.2 Site Checksheet

used as an indicator of recent channel improvement works which could affect the biota.

Further observations recorded for each sampling station included distance from the outfall; whether the station was in light or shade; the aquatic plants present; and the nature of the substratum. Ammonia levels were added to the checksheets following an offer of the use of an ammonia probe, but were not measured because the equipment was not available at the time of sampling. It had been hoped that ammonia would provide a guide to the extent of recent sewage pollution.

The nature of the substratum was recorded in terms of percentages of stones, pebbles and sand or finer material. The proportions were estimated by visual inspection. The substratum and aquatic plant data were only used in subsequent analysis if there were large differences between stations at the same site.

5.6 DETERMINATION OF CATCHMENT CHARACTERISTICS

Information relating to the study catchments was collected and recorded on Site Information Forms (Fig. 5.3).

Details of the location of each site, access to the sampling stations, the relevant local authority and the Severn-Trent division were recorded. The person who had suggested the site was noted as a source of additional information. A record was also kept of dates of visits and sampling undertaken at each site.

The remaining items on the form are characteristics of the sewered and receiving stream catchments themselves, obtained from local authority and Severn-Trent records.

Whenever possible, sewered catchment areas were obtained directly from local authorities, but in most cases they had to be

SITE REF ________

Name:

Watercourse:

NGR:

STWA Division:

County:

Local Authority:

Catchment Area:

Land Use:

% Impervious:

Outfall Diameter:

NWC Class:

Upstream Catchment Area:

% of Upstream Catchment Urban:

Drainage Area Ratio:

Site Suggested By:

Access:

Visits:

Photograph Nos:

Comments:

Fig. 5.3 Site Information Form

estimated from maps and plans using a planimeter. Copies of the drainage plans for each catchment were obtained from the appropriate local authority, and the catchment boundaries marked for this purpose.

Plans were also used to classify catchment land use into one of the following categories: residential, industrial, commercial, highway, or open. The proportion of each catchment occupied by each land use class was calculated from areas determined by planimetry.

The percentage of the catchment covered by impervious surfaces was obtained in most cases from the local authorities, and estimated from plans where not available.

The outfall diameter on the sewer records was recorded for comparison with the diameter measured on site.

The catchment area of the river above each outfall was estimated from 1:50 000 and 1:25 000 OS maps using a planimeter and Severn-Trent subcatchment plans. The proportion of this upstream area covered by urban development was also estimated from these maps. The upstream catchment area was used to calculate the drainage area ratio (DAR) for each site, where

$$DAR = \frac{\text{Urban Area Contributing Runoff}}{\text{Stream Drainage Area Upstream of Urban Input}} \times 100$$

DAR was used in the Nationwide Urban Runoff Program (US EPA, 1983b) and is a measure of the location of the urban area relative to the headwaters of the receiving streams.

The NWC quality class of the river at each site was obtained from Severn-Trent records (STWA, 1987).

Local authority drainage records, used to obtain many of the recorded characteristics, were not always accurate and were often poorly maintained. The use of a planimeter to measure areas is also not particularly accurate, especially for 1:50 000 maps. Catchment characteristics measured from plans and maps were therefore regarded as estimates which could be used to group catchments into broad classes but could not be used for detailed statistical analysis.

The methods used to analyse and interpret the data are described in Chapter 6.

CHAPTER 6

DATA ANALYSIS METHODS

The procedures in this chapter are those used for analysis of the data described in chapter 5. Procedures covered include the overall analysis strategy adopted, general data organisation, the treatment of biological data and biological scores, and the application of statistical tests.

6.1 ANALYSIS STRATEGY AND DATA ORGANISATION

Data were grouped into three categories: catchment characteristics, biological data, and metals data. Each category was further divided into sub-categories which contained data items. The heirarchical system is illustrated in Table 6.1.

The overall strategy adopted in analysis of these data was as follows:

a) Describe the data in each sub-category using tables and graphs with descriptive statistics for each item, e.g. describe the range of BMWP scores at upstream stations.

b) Investigate relationships between data items in the same sub-category, e.g. correlation between zinc and cadmium in algae samples.

c) Investigate relationships between data items in the same category but different sub-categories, e.g. correlation between zinc in algae and zinc in sediment samples.

TABLE 6.1 DATA ORGANISATION

CATCHMENT CHARACTERISTICS	BIOLOGICAL DATA	METALS DATA
SITES	BIOL	ALGMET1, SEDMET1
catchment area/land use/ impervious surfaces/DAR/ % upstream catchment urban	BMWP/WBMWP scores + taxa counts at each station	Zn/Cd/Pb/Ni/B/Cr in algae/sediment at U + D stations
DSCHG	COEFFS	ALGMET2, SEDMET2
discharge descriptions for each site	Jaccard's coeff./ Kothe's Species Deficit	V/Be/Cu/Ag/As/Se in algae/sediment at U + D stations
TSITES	TBIOL	TALGMET, TSEDMET
transformed SITES variables	transformed BIOL variables	transformed ALGMET and SEDMET variables
	DBIOL	DALGMET, DSEDMET
	station-to-station differences in BIOL variables	upstream/ downstream differences in ALGMET and SEDMET variables

d) Investigate relationships between data items in different categories, e.g. metals in sediments and BMWP scores at upstream stations.

e) Investigate upstream/downstream differences in metals and biological categories.

f) Relate upstream/downstream differences to catchment characteristics.

The data were stored on disk, in files corresponding to the sub-categories in Table 6.1. Additional files were created for transformed data and station-to-station differences in metals and biological parameters. Raw data files were checked against data sources for errors, and then frozen. Transformations and differences were generated from the raw data to ensure that the integrity of the data set was maintained. Analysis of the digitised data was carried out on an Apricot PC using the ABSTAT statistical package and SUPERCALC spreadsheets.

6.2 CALCULATION OF BIOLOGICAL PARAMETERS

The biological indices used were BMWP scores, number of taxa present, weighted BMWP (WBMWP) scores, Jaccard's coefficient and a modified version of Kothe's Species Deficit.

Weighted BMWP scores were used because an index which took abundance of organisms into account was required. Alternative measures of abundance such as total number of individuals and mean or median number per taxon were considered, but rejected because large numbers of pollution-tolerant organisms were not distinguished from large numbers of intolerant organisms. The use of a weighted BMWP score crudely permits this distinction.

Jaccard's coefficient was used as an index of affinity based on the joint presence of taxa. Kothe's Species Deficit, which was not used in the pilot study, was used in a modified form which measures the suppression of upstream taxa.

BMWP and WBMWP scores were calculated by hand for all the samples collected. Jaccard's coefficient and Kothe's Species Deficit were calculated from taxa counts using ABSTAT.

6.2.1 BMWP and WBMWP scores

BMWP scores were derived by preparing a list of families present, allocating points to each using a "Score Card" (Table 6.2) and summing the points to give the score. Table 6.3 includes an example of BMWP score calculation.

Following discussions with hydrobiologists at Aston University and Middlesex Polytechnic, the calculation of WBMWP scores described in chapter 4 was simplified. Scores were calculated by multiplying scores and factors for each taxon and summing them to derive the weighted score:

WBMWP SCORE = SUM (SCORES x FACTORS)

This had the disadvantage that weighted scores no longer approximated to true BMWP scores, but the advantage that differences were more easily detected.

The weighting system was also altered, to dampen its effect slightly. The new weighting system was as follows:

NO. OF INDIVIDUALS	DESCRIPTION	WEIGHTING FACTOR
1 - 2	Present	1
3 - 10	Few	1.5
11 - 50	Common	2
51 - 100	Abundant	2.5
> 100	Very abundant	3

FAMILY	SCORE	FAMILY	SCORE	FAMILY	SCORE	FAMILY	SCORE
Planariidae	5	Siphlonuridae	10	Mesovelidae	5	Rhyacophilidae	7
Dendrocoelidae	5	Baetidae	4	Hydrometridae	5	Philopotamidae	8
		Heptageniidae	10	Veliidae	5	Polycentropidae	7
Neritidae	6	Leptophlebidae	10	Gerridae	5	Psychomiidae	8
Viviparidae	6	Ephemerellidae	10	Nepidae	5	Hydropsychidae	5
Valvatidae	3	Potomanthidae	10	Nauccordae	5	Hydroptilidae	6
Hydrobiidae	3	Ephemeridae	10	Aphelochheiridae	10	Phryganeidae	10
Lymnaeidae	3	Caenidae	7	Notonectidae	5	Limnephilidae	7
Physidae	3	Caenidae	7	Pleidae	5	Molannidae	10
Planorbidae	3	Taeniopterigidae	10	Corixidae	5	Beraeidae	10
Ancylidae	6	Leuctridae	10			Odontoceridae	10
Unionidae	6	Nemouridae	7	Haliplidae	5	Leptoceridae	10
Sphaeriidae	3	Capniidae	10	Hygrobiidae	5	Goeridae	10
		Perlodidae	10	Dytiscidae	5	Leptostomatidae	10
(Oligochaeta)	1	Perlidae	10	Gyrinidae	5	Branchycentridae	10
		Chloroperlidae	10	Hydrophildae	5	Sericostomatidae	10
Piscicolidae	4			Clambidae	5		
Glossiphonidae	3	Platycnemididae	6	Helodidae	5	Tipulidae	5
Erpobdellidae	3	Coenagriidae	6	Dryopidae	5	Chironomidae	2
		Lestidae	8	Elminthidae	5	Simulidae	5
Asellidae	3	Agriidae	8	Chrysomelidae	5		
Corophiidae	6	Gomphidae	8	Curculionidae	5		
Gammaridae	6	Cordulegasleridae	8				
		Aeshnidae	8	Sialidae	4		
		Corduliidae	8				
		Libellulidae	8				

TABLE 6.2 BMWP SCORE CARD

The example of WBMWP score derivation given in Table 4.2 is repeated in Table 6.3 using the revised procedure, to demonstrate how the weighting factors reflect organism abundance.

TABLE 6.3 BMWP AND REVISED WBMWP SCORE DERIVATION

	COUNT IN SAMPLE		WEIGHTING FACTOR		BMWP SCORE	SCORE x FACTOR	
	U	D	U	D		U	D
Ancylidae	3	1	1.5	1	6	9	6
Sphaeriidae	3	1	1.5	1	3	4.5	1
Oligochaeta	42	14	2	2	1	2	2
Glossiphonidae	3	1	1.5	1	3	4.5	3
Baetidae	9	3	1.5	1.5	4	6	6
Ephemerellidae	6	2	1.5	1	10	15	10
Hydropsychidae	3	1	1.5	1	5	7.5	5
Chironomidae	18	6	2	1.5	2	4	3
Gammaridae	240	80	3	2.5	6	18	15
					40	70.5	51

BMWP SCORE = 40
WBMWP (UPSTREAM) = 70.5
WBMWP (DOWNSTREAM) = 51

6.2.2 Jaccard's coefficient

This was calculated from the equation introduced in section 4.1.2:

$$J = \frac{c}{a + b - c}$$

where a = no. of taxa at upstream station

b = no. of taxa at downstream station

c = no. of taxa common to both stations

Jaccard's coefficient measures similarity, with a coefficient of 1.00 indicating identical taxa lists at the two stations.

6.2.3 Kothe's Species Deficit

Kothe's index as described in chapter 4 is based on the observation that pollution often causes a reduction in the number of taxa below an effluent outfall. It is expressed as a percentage, where a value of 100% indicates complete suppression of the fauna, but this is only true if disappearing taxa are not replaced by others. Hellawell (1978) suggested a modified index which measures suppression of upstream fauna:

$$DM = \frac{A - M}{A} \times 100$$

where A = no. of taxa present at the upstream station
M = no. of these taxa missing at the downstream station

A value of 100% means that none of the upstream taxa disappear below the outfall and a value of 0% means complete suppression of the upstream fauna. Kothe's Species Deficit was used in this modified form.

6.3 STATISTICAL ANALYSIS

Relationships between data items were investigated using correlation analysis and, for grouped data, analysis of variance (anova). To test the significance of upstream/downstream differences, paired t-tests were used. The effects of catchment characteristics on outfall impact were assessed by anova and t-tests.

The details of these tests are well documented elsewhere (e.g Sokal and Rohlf, 1969; Elliott, 1973; Norcliffe, 1982) and the reader is referred to these texts for full details. The reasons for selecting the tests, the assumptions underlying them, and how data were tested to ensure these assumptions were met are described below.

A significance level of 10% was selected for the tests used. This level was chosen so that type 2 errors were avoided, and significant relationships were not missed. The possibility of making type 1 errors - concluding that an outfall was damaging to water quality when it was not, or finding significant relationships between variables when none existed - was considered preferable to erroneously dismissing such effects as insignificant. Significance levels higher than 10% are reported if they apply. The statistical tables used were those provided by Lindley and Scott (1984).

6.3.1 Correlation coefficients

Correlation analysis was chosen rather than regression analysis because the required information was a measure of the association between data items. The form of relationships between data items was not required, and data collection had not been designed to facilitate regression analysis.

The correlation coefficient, r, is only meaningful when the pair of variables under investigation follow the bivariate normal distribution. The distribution of data was therefore tested before analysis, by inspection of histograms and calculation of the sample statistics for skewness and kurtosis. Data which were not normally distributed were transformed using logs and square roots.

6.3.2 t-tests and anova

Paired t-tests were used to test the significance of observed differences in upstream and downstream parameters. Single classification anova was used to assess the effect of catchment characteristics on upstream/downstream differences, by grouping sites according to land use, area, and other characteristics.

The t-test for the difference between two means is mathematically equivalent to a single classification (one-way) analysis of variance with two groups. The t-test for paired comparisons is equivalent to a two-way anova for paired comparisons. The assumptions of anova and t-tests are that samples are independent, random, homoscedastic and normally distributed.

The assumption of randomness was assumed to have been met by the method of site selection, as sites were selected at random from the population of possible sites which satisfied the selection criteria. Independence in a non-experimental situation such as this cannot be guaranteed, as the impact of one outfall could influence the effect of another further downstream. It was assumed that this was not the case, and although it is possible that upstream quality influences impact, it was assumed that upstream quality was not a function of the impact of another outfall included in the study. This assumption is only problematic for the few sites which were close together.

The assumptions of normality and homoscedasticity (homogeneity of variances) were tested for their validity. Normality was tested as described in section 6.3.1, but as anova is robust with respect to deviations from normality, only very skewed distributions were subjected to transformations.

Where there were two groups of data to be tested for homoscedasticity, an F-test was used for the hypothesis that the sample variances were equal. For more than two groups, Bartlett's test for homogeneity of variances was applied. This test is also sensitive to departures from normality, so samples found to be homoscedastic using this test are also probably normally distributed. Moderate heterogeneity of variances does not seriously affect the significance test at the end of an anova, but in severe cases transformations were carried out.

CHAPTER 7

SURVEY RESULTS

The purpose of this chapter is to present the results of the survey work, including the site selection programme and location of sites. Analysis, interpretation and discussion of the results are covered in chapters 8 to 10. Results are tabulated and summarised in three sections: sites, biological parameters, and metals in sediments and algae.

7.1 STUDY SITES

7.1.1 Site selection

It was estimated that it would be possible to sample 50 sites between June and October 1987, and that at least half the sites visited would not satisfy the criteria for inclusion in the project (section 5.1). In practice, the rejection rate was higher than 50% and more site visits than had been anticipated were necessary.

Twelve local drainage authorities and three Severn-Trent Water Divisions were contacted and asked to assist in the identification of candidate sites. Approximately 200 potential sites were identified, all of which appeared to satisfy the selection criteria, although the suitability of watercourses for cylinder sampling could not be assessed without a site visit.

145 sites (including 12 from the pilot study) were visited and 49 were selected for sampling. The reasons why the remaining sites were rejected are summarised in Table 7.1.

TABLE 7.1 SITE SELECTION - SUMMARY OF OUTFALL SITES VISITED

RESULT OF VISIT	No. of sites
SITE SELECTED : criteria satisfied	49
SITE REJECTED : insufficient flow rate, too shallow, no riffle zones, unsuitable for cylinder sampling	37
water too deep to enter safely	11
no access	10
excessive weed growth blocking channel	10
culverted sections preventing sampling at one or more stations	7
artificial channel	6
artificially reinforced bed	3
river reach severely polluted by other source	4
lake immediately upstream or downstream	2
weir immediately upstream or downstream	3
confluence with second watercourse immediately upstream or downstream	3
TOTAL REJECTED	96

NOTE : 2 of the selected sites were subsequently rejected because the catchments were impossible to define from available plans.

The most common reason for rejection was that the watercourse was too slow flowing, too shallow or otherwise physically unsuitable for cylinder sampling. Other problems encountered several times were poor access, deep water and excessive weed growth blocking the channel - none of which could be assessed from maps and plans.

Two of the selected sites were rejected after they had been sampled because it was impossible to define the catchments using the available plans. The number of sites for which results are quoted is therefore 47.

7.1.2 Location of sites

The 47 study sites were all in the Severn-Trent region and distributed between 11 sub-catchments of the Severn and Trent River Basins. The location of sites is illustrated in Fig. 7.1, and a list of sites with reference numbers and precise outfall locations given in Table 7.2.

TABLE 7.2 SITES STUDIED AND OUTFALL LOCATIONS

CATCHMENT	WATERCOURSE	NO.	SITE NAME	NGR
UPPER BLYTHE	R Blythe	1	Cheswick Gn	SP 126 754
	R Blythe	2	Stratford Rd	SP 145 758
R COLE	R Cole	3	Trueman's Hth	SP 099 773
	R Cole	4	Corley Cl	SP 103 783
	R Cole	5	Green Ln	SP 103 784
	R Cole	6	Colebrook Rd	SP 103 788
	R Cole	7	Atherstone Cl	SP 103 789
	R Cole	8	Robin Hood Ln	SP 098 812
	R Cole	9	Cole Bank Rd	SP 100 818
	R Cole	10	Green Rd Ford	SP 100 821
	R Cole	11	Cubley Rd	SP 100 823
	R Cole	12	Babb's Mill	SP 161 877
	R Cole	13	Cook's Ln	SP 175 878
	Hatchford Brook	14	Elmdon Park	SP 159 834
	Hatchford Brook	15	Airport	SP 164 855
	Kingshurst Brook	16	Bosworth Dr	SP 175 868
	Kingshurst Brook	17	Chelmsley Wood	SP 175 870
	R Cole	18	Chester Rd	SP 183 874

TABLE 7.2 continued

CATCHMENT	WATERCOURSE	NO.	SITE NAME	NGR
R REA	Callow Brook	19	The Avenue	SO 985 774
	Griffins Brook	20	Griffin Cl	SP 032 810
	Griffins Brook	21	Bristol Rd	SP 032 810
	Griffins Brook	22	Alder Lane A	SP 032 810
	Griffins Brook	23	Alder Ln B	SP 032 810
	Bourn Brook	24	Leasow Dr A	SP 039 832
	Bourn Brook	25	Leasow Dr B	SP 041 832
	Bourn Brook	26	Leasow Dr C	SP 042 832
R TAME	Plants Brook	27	Reddicapp	SP 128 954
UPPER SOAR	Whetstone Brook	28	Ashville Way	SP 557 965
	Whetstone Brook	29	Bridge Farm	SP 558 968
R BLITHE	R Blithe	30	Willowdale	SJ 951 414
	R Blithe	31	Blythe Bdge Stn	SJ 956 411
R PENK	Trib of Penk	32	Little Marsh	SJ 932 148
R SALWARPE	Battlefield Brook	33	Catshill	SO 957 736
	Battlefield Brook	34	Whitford Rd	SO 949 706
	Battlefield Brook	35	Deans Way	SO 950 706
	Spadesbourne Brook	36	Brook Ln	SO 954 702
	Spadesbourne Brook	37	Worcester Rd	SO 954 700
	Spadesbourne Brook	38	Lyttleton Ave	SO 955 697
	Spadesbourne Brook	39	Charford Rd	SO 956 697
	Sugar Brook	40	Stoke Rd	SO 961 694
BOW BROOK	Wharrage Brook	41	Hunt End	SP 032 637
R STOUR (WORCS)	Smestow Brook	42	Fowler Park	SJ 919 001
	Smestow Brook	43	Trysull	SO 856 938
	Smestow Brook	44	Wombourne	SO 857 914
R SOWE	R Sowe	45	Woodway Park	SP 375 811
	Pickford Brook	46	Woodridge Ave	SP 292 805
	Trib of Sherbourne	47	Risborough Cl	SP 302 794

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Fig 7.1 Site Locations

7.1.3 Catchment characteristics

General site characteristics and catchment land use are given in Tables 7.3 and 7.4.

The study catchments ranged in size from a third of a hectare to 500 hectares, with a median area of 9.35 hectares. 59% of the catchment areas were between 1 and 20 hectares, but small and large catchments were both well represented, with 8 under 1 hectare and 8 over 50 hectares.

The drainage area ratio, DAR, (see section 5.6) is a measure of the size of the sewered catchment relative to the receiving stream, and can be used as a guide to the contribution to flow made by the sewered area. For most of the sites (72%) the DAR was less than 1, and only 3 sites had a DAR over 5. This was as expected: high DARs indicate large urban areas discharging into small streams at a single point, whereas in practice large urban areas are drained through several outfalls.

The degree of urbanisation of the river catchment upstream of each outfall was estimated as a percentage of the total catchment area. 6 outfalls were at points where the entire upstream catchment was urban, and 5 were at points where urbanisation upstream was less than 10%. The majority of the outfalls were at points where the degree of urbanisation upstream was between 20 and 80%.

All the receiving streams sampled were NWC class 1B or class 2 in 1987.

TABLE 7.3 GENERAL CATCHMENT CHARACTERISTICS

SITE NO.	AREA (ha)	IMPERVIOUS %	DAR	% UPSTREAM AREA URBAN	NWC CLASS
1	5.31	30	0.26	9.4	2
2	5.3	80	0.17	6.4	2
3	0.6	100	0.04	4.3	1B
4	2.4	30	0.10	16.4	1B
5	4.4	30	0.19	16.5	1B
6	11.3	30	0.46	20.0	1B
7	0.73	30	0.03	20.4	1B
8	45	30	1.10	47.1	1B
9	11.35	30	0.26	50.6	1B
10	9.26	30	0.20	52.4	1B
11	9.35	30	0.21	52.5	1B
12	147	30	1.91	71.8	2
13	10.5	35	0.13	73.7	2
14	14.5	30	2.47	100.0	-
15	19	75	0.71	100.0	-
16	2.2	50	0.05	89.2	2
17	66.75	40	1.60	89.2	2
18	56	35	0.44	79.4	2
19	3.375	80	4.69	51.4	-
20	1.95	50	0.37	66.8	1B
21	19.4	41	3.67	66.9	1B
22	0.67	40	0.12	68.1	1B
23	0.98	40	0.18	68.1	1B
24	0.31	100	0.02	100.0	1B
25	5.4	30	0.31	100.0	1B
26	73.6	30	4.2	100.0	1B
27	500	28	25.6	55.1	1B
28	14.71	78	0.73	45.0	-
29	4.69	30	0.22	5.7	-
30	115.5	40	2.46	20.0	1B
31	12.5	30	0.63	23.3	1B
32	7.2	30	0.60	0.0	-
33	5.55	40	0.41	16.3	-
34	0.5	100	0.13	18.6	2
35	9.75	85	0.48	19.9	2
36	2	90	0.06	26.7	1B
37	0.92	100	0.03	26.8	2
38	11.5	45	0.36	27.9	2
39	9.5	45	0.29	29.1	2
40	4	72	0.12	30.8	2
41	27.1	35	8.71	31.5	-
42	58	60	19.90	100.0	-
43	0.836	58	0.02	39.5	2
44	20.5	90	0.27	38.0	2
45	120	20	2.24	51.9	2
46	5.15	40	0.76	58.0	2
47	14.5	30	2.99	63.9	-

TABLE 7.4 CATCHMENT LAND USE

SITE NO.	RESIDENTIAL %	INDUSTRIAL %	COMMERCIAL %	HIGHWAY %	OPEN %
1	100	0	0	0	0
2	0	0	57	43	0
3	0	0	0	100	0
4	100	0	0	0	0
5	100	0	0	0	0
6	100	0	0	0	0
7	100	0	0	0	0
8	94	6	0	0	0
9	100	0	0	0	0
10	100	0	0	0	0
11	93	7	0	0	0
12	52	28	7	0	13
13	100	0	0	0	0
14	100	0	0	0	0
15	0	100	0	0	0
16	1	0	0	99	0
17	91	0	9	0	0
18	79	2	5	0	14
19	0	100	0	0	0
20	100	0	0	0	0
21	62	0	0	38	0
22	100	0	0	0	0
23	100	0	0	0	0
24	0	0	0	100	0
25	100	0	0	0	0
26	56	11	16	0	17
27	69	1	8	0	22
28	0	56	0	44	0
29	100	0	0	0	0
30	58	33	0	0	9
31	100	0	0	0	0
32	100	0	0	0	0
33	86	0	0	14	0
34	0	0	0	100	0
35	100	0	0	0	0
36	0	100	0	0	0
37	0	0	0	100	0
38	100	0	0	0	0
39	100	0	0	0	0
40	50	0	0	50	0
41	51	0	0	0	49
42	50	50	0	0	0
43	0	0	60	40	0
44	0	100	0	0	0
45	70	0	10	0	20
46	100	0	0	0	0
47	100	0	0	0	0

Most of the study catchments were residential. 45% were completely residential, and a further 13% were mainly (70% or more) residential. Other land uses were also represented - large catchments were usually mixed land use, and although there were no entirely commercial sites, 4 catchments had commercial areas which accounted for over 10% of the land use.

Industrial catchments were well represented: 4 sites were entirely industrial, and another 9 contained significant industrial areas. This is not immediately apparent from the figures in Table 7.4, as the industrial proportions of mixed catchments were small; but the mixed catchments tended to be large. The airport site monitored was classed as an industrial catchment.

The highway land use category is not as well defined as the others and merits further explanation. Areas classed as highways were major roads, and streets in areas where buildings drained to a different system or to soakaways. Most of the sites quoted as highway, especially those with a small area, were major roads.

A few of the sites have areas classed as open. These are almost all city park areas.

7.1.4 Field observations

The only field observations used for all the sites in the data analysis were those of discharges from outfalls. Other field observations are described briefly before discharges are considered. A selection of photographs of the study sites, and some of the sites visited which were unsuitable for sampling, is included at the end of this chapter.

No significant scour effects were observed at outfalls. There was usually some silting immediately below the outfall, and the substratum below an outfall was generally richer in fine sediment than the substratum above an outfall. Visible silting always appeared to be confined to the outfall station, but the extent was often difficult to assess and may have been more widespread. As the degree of silting apparent at any time was dependent on flow conditions and no extreme cases were observed, this was not considered further.

The water plants present at upstream and downstream stations were generally the same. There was one clear exception: at the airport site (no. 15), where the runoff was from a runway, the river downstream of the outfall was completely overgrown with weeds, particularly watercress. This was attributed to increased nutrients, derived from de-icing salts used on the runway; a problem already identified by the local authority at other sites receiving airport runoff.

Discharge observations were made in both wet and dry conditions. 10 outfalls were visited in wet weather, and 9 were operating: 2 of the observed discharges were contaminated with foul sewage; 2 appeared clean; and the other 5 were turbid and oily but there was no evidence of foul sewage.

37 outfall sites were visited in dry weather, and 26 were operating. At one of the dry outfalls there was visible evidence of a recent foul sewage discharge. 10 of the dry weather discharges contained sewage, 2 were industrial effluent, and for 5 the effluent sources were not identified but the discharges were not clean. The other 9 outfalls operating in dry weather were discharging apparently clean water, which was assumed to be infiltrating from leaking water mains or groundwater and not damaging to river quality.

A summary of the discharge information is presented in Table 7.5.

TABLE 7.5 OBSERVED DISCHARGES AT OUTFALL SITES

DESCRIPTION	NO.OF SITES	% OF SITES
No evidence of contamination or illegal discharges	27	57.4
Sewage contamination	13	27.7
Industrial discharge in dry weather	2	4.3
Unidentified discharge in dry weather (not clean)	5	10.6
TOTAL	47	100

7.2 BIOLOGICAL PARAMETERS

7.2.1 Taxa counts and BMWP scores

Taxa counts, BMWP and weighted BMWP (WBMWP) scores for the three sampling stations at each site are given in Table 7.6, with summary statistics in Table 7.7 and Fig. 7.2.

BMWP scores ranged from 7 (unsatisfactory) to 72 (good) where the description is the "General Biological Description" applied regionally by Severn-Trent Water as an aid to interpretation:

BMWP Score	0-9	Unsatisfactory	51-100	Good
	10-25	Poor	101-150	Very good
	26-50	Moderate	151+	Excellent

Most scores were in the range 26-50, indicating streams of moderate quality.

Overall, there was little difference between upstream, outfall and downstream stations in mean or median taxa count or BMWP score, although differences at individual sites were often considerable. For taxa counts, BMWP, and WBMWP scores, the upstream mean and median values for all sites were marginally higher than the downstream values. Station-to-station differences are investigated fully in Chapter 9.

7.2.2 Comparative indices

Values of Kothe's Species Deficit (DM) and Jaccard's coefficient (J) for comparison of taxa present at upstream and downstream stations are given in Table 7.8.

The highest value of DM is 100, indicating that all the taxa present at the upstream station are also present downstream of the outfall. Where this coincides with a high value of J, few additional taxa are present at the downstream station, and the taxa lists are similar. A high value of DM and a low value for J indicates that taxa are present at the downstream station which were not present upstream.

The lowest value of DM is 45.45, indicating that 55% of the taxa present upstream disappeared downstream of the outfall. Low values of DM are always reflected by low values of J.

For sites where DM is exactly 100 times J, all the taxa at the downstream station were present at the upstream station.

A more detailed investigation of the relationships between the biological indices used is included in Chapter 8.

TABLE 7.6 TAXA COUNTS AND BMWP SCORES AT UPSTREAM, OUTFALL AND DOWNSTREAM STATIONS

SITE	TAXA COUNT			BMWP SCORE			WBMWP SCORE		
NO.	U	O	D	U	O	D	U	O	D
1	11	8	6	47	32	21	72	49	29.5
2	6	12	14	28	57	61	43	78.5	92
3	8	8	7	35	35	28	56	61	53
4	8	6	11	25	15	44	32.5	19.5	56
5	11	9	5	44	30	18	56	48	27.5
6	9	6	10	39	18	43	59	23.5	70.5
7	10	10	8	43	43	34	70.5	60.5	51.5
8	10	12	10	37	43	35	64	68.5	61.5
9	9	10	7	38	43	21	67	50	40
10	9	11	9	31	40	28	46	65	40.5
11	7	9	8	21	29	24	45	52.5	47
12	5	5	3	12	14	7	23	19	10.5
13	3	5	6	6	12	16	10	18.5	33
14	9	10	10	30	35	31	48.5	46.5	48.5
15	11	8	8	41	26	25	66.5	40.5	36.5
16	5	6	8	15	18	25	26.5	31.5	37
17	8	7	6	25	21	18	37	39	25
18	8	8	7	24	24	18	37	38.5	29.5
19	5	5	6	16	16	18	24.5	23.5	22
20	8	7	7	35	28	33	54.5	39.5	50
21	7	8	6	33	33	25	50	51	42
22	6	8	10	25	33	41	42	50	56.5
23	10	7	9	41	24	38	56.5	29	51.5
24	7	7	6	22	22	19	38	41	39.5
25	6	4	8	19	12	27	39.5	28	47
26	8	6	7	27	19	25	47	31.5	37
27	11	8	10	48	38	44	88.5	61	53
28	10	7	9	38	23	29	55	30	40.5
29	9	3	7	29	6	21	38	7.5	24
30	14	12	12	72	62	53	92.5	74	73
31	11	6	12	52	19	60	68.5	29	81.5
32	10	13	10	37	60	43	64	82	51.5
33	7	5	6	26	19	21	45	37.5	32.5
34	7	9	4	25	37	17	46	53	38
35	6	6	6	18	21	21	27	33	33
36	8	7	6	29	25	21	43	45	40
37	6	4	6	21	13	21	40	23.5	42
38	6	7	7	21	31	33	40.5	47.5	52.5
39	6	8	6	21	31	21	38	46	33
40	8	6	6	26	22	24	47	36	39.5
41	5	3	4	23	12	15	44.5	25.5	33
42	4	4	4	10	9	9	20.5	16	12
43	6	5	5	22	18	18	32.5	38	35
44	6	4	4	21	12	10	37.5	15.5	15
45	9	8	7	31	26	23	51.5	43	44.5
46	9	9	6	32	36	19	54	60.5	36.5
47	7	8	7	22	27	22	39.5	46	39

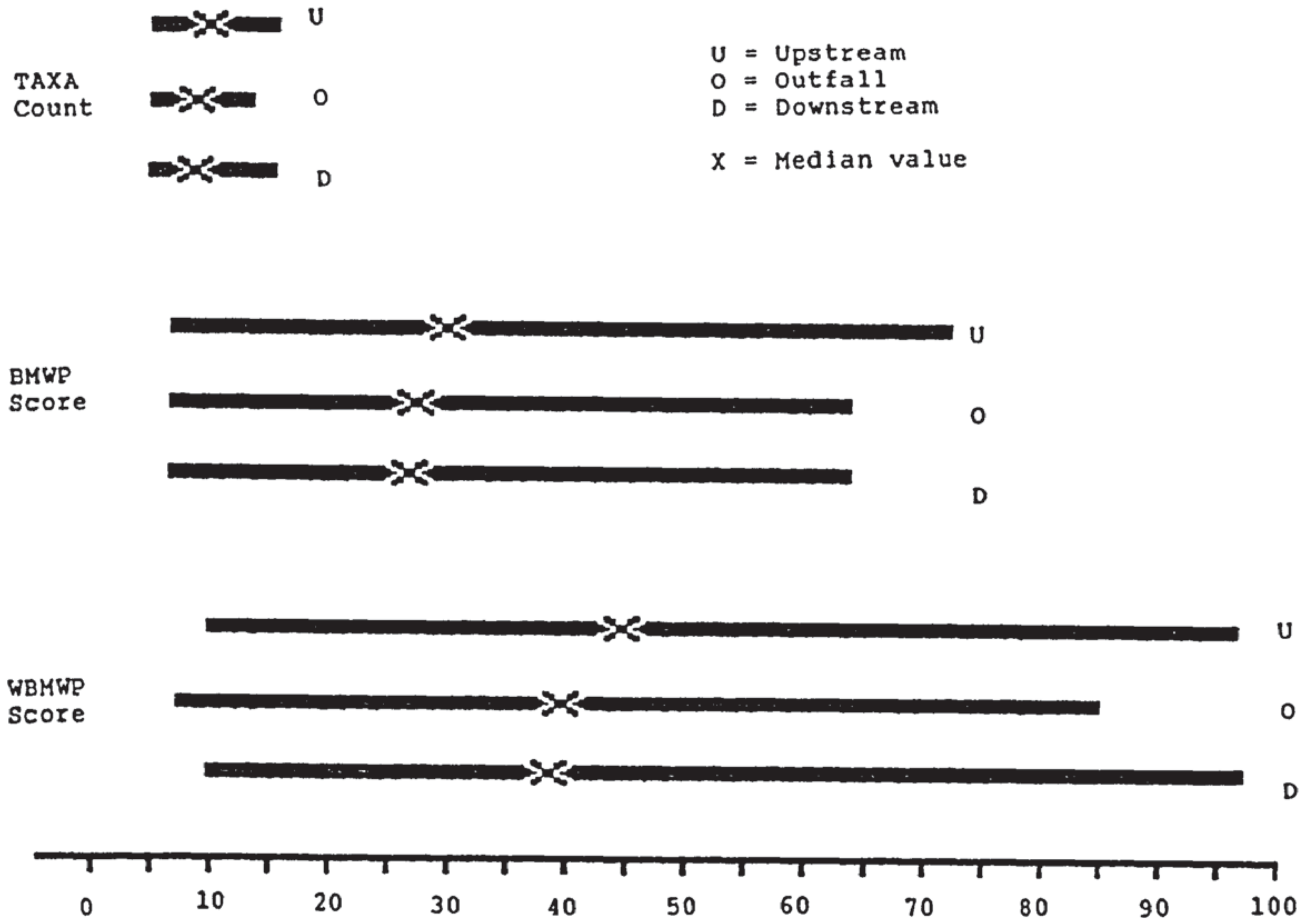


Fig. 7.2 Ranges of TAXA Counts and BMWP Scores

TABLE 7.7 SUMMARY OF TAXA COUNTS AND BMWP SCORES

	MEAN	MEDIAN	STD.DEV.	MIN	MAX
TAXA COUNT					
upstream	7.85	8	2.22	3	14
outfall	7.32	7	2.42	3	13
downstream	7.36	7	2.34	3	14
BMWP SCORE					
upstream	29.43	27	12.01	6	72
outfall	27.00	25	12.87	6	62
downstream	26.98	24	12.17	7	61
WBMWP SCORE					
upstream	47.33	45	16.54	10	92.5
outfall	41.54	40.5	17.28	7.5	82
downstream	42.20	40	16.28	10.5	92

TABLE 7.8 COMPARATIVE INDICES FOR UPSTREAM AND DOWNSTREAM STATIONS

SITE NO.	KOTHE DM	JACCARD J	SITE NO.	KOTHE DM	JACCARD J
1	45	0.42	25	100	0.75
2	100	0.43	26	75	0.67
3	75	0.67	27	63	0.50
4	100	0.73	28	70	0.58
5	45	0.45	29	78	0.78
6	78	0.58	30	79	0.73
7	80	0.80	31	73	0.53
8	90	0.82	32	60	0.43
9	67	0.60	33	71	0.63
10	78	0.64	34	57	0.57
11	100	0.88	35	83	0.71
12	60	0.60	36	63	0.56
13	67	0.29	37	100	1.00
14	89	0.73	38	83	0.63
15	55	0.46	39	83	0.71
16	100	0.63	40	63	0.56
17	63	0.56	41	60	0.50
18	75	0.67	42	75	0.60
19	80	0.57	43	83	0.83
20	88	0.88	44	67	0.67
21	71	0.63	45	67	0.60
22	100	0.60	46	67	0.67
23	80	0.73	47	100	1.00
24	86	0.86			

7.3 METALS IN SEDIMENTS AND ALGAE

7.3.1 Metals in sediments

Sediment samples were taken at 45 of the 47 sites monitored. A statistical summary of the metals concentrations found is given in Table 7.9, with the full results in Table 7.10.

TABLE 7.9 SUMMARY OF METALS IN SEDIMENT SAMPLES

All concentrations as mg/kg dry sediment

		MEAN	MEDIAN	STD.DEV	MIN	MAX	DF %
Zinc	U	204.0	95	429.3	LT 1	2800	97.8
	D	170.6	99	238.6	LT 1	1300	97.8
Cadmium	U	-	LT 0.5	-	LT 0.5	13.2	26.7
	D	-	LT 0.5	-	LT 0.5	24.0	20.0
Lead	U	218.3	180	140.3	78	950	100
	D	206.6	180	87.87	95	540	100
Nickel	U	68.27	54	49.21	12	310	100
	D	63.56	53	30.69	12	130	100
Boron	U	26.86	19	32.73	LT 1	180	95.6
	D	24.19	17	25.97	LT 1	140	93.3
Chromium	U	14.13	10	15.67	LT 1	86	97.8
	D	12.89	11	12.18	1	81	100
Vanadium	U	14.62	13	9.148	LT 1	56	97.8
	D	14.82	13	9.568	LT 1	46	97.8
Copper	U	64.53	18	240.3	LT 1	1600	91.1
	D	37.24	16	66.73	LT 1	340	95.6
Silver	U	-	LT 1	-	LT 1	9.9	8.9
	D	-	LT 1	-	LT 1	31.6	15.6
Arsenic	U	2.456	2.4	2.071	LT 1	15	97.8
	D	2.647	2.2	2.258	LT 1	14	97.8

DF = Detection frequency; U = upstream; D = downstream

For metals which were detected in all samples, or all but one or two, the summary statistics were calculated on the whole data set with zero substituted for 'not detected'. For cadmium and silver, which were not always present, the mean and standard deviation are not quoted. Selenium was not detected in any of the samples, and beryllium in only one sample. These elements are therefore not considered further.

As a quick means of assessing the level of metal enrichment in sediments, and in the absence of any other background values, Forstner and Wittmann (1981) suggest comparison with a 'shale standard'. The relevant metals concentrations (mg/l) in average shale are:

zinc	95	chromium	90
cadmium	0.3	vanadium	130
lead	20	copper	45
nickel	68	silver	0.27
boron	100	arsenic	13

Comparison with the maximum values in Table 7.9 shows that in some samples there was considerable enrichment for zinc, cadmium, lead, nickel, boron, vanadium, copper and silver. For zinc and lead - elements which are often found in higher concentrations in urban areas - this enrichment was apparent in most samples, as indicated by the mean and median concentrations. Copper, which is also commonly enriched by man's activities, was not generally found in high concentrations, although high values were recorded for the Bourn Brook in Birmingham.

One of the problems of comparing metals concentrations measured by different studies is that different analytical methods, particularly extraction techniques, can significantly affect the results. Inter-study comparisons are therefore only valid if the same methods have been used. Taking this into consideration, the lead and zinc concentrations in Tables 7.9 and 7.10 were compared with values quoted by Forstner and Wittmann for sediments in West

TABLE 7.10 METAL CONCENTRATIONS IN SEDIMENT SAMPLES

All concentrations in mg/kg dry sediment

SITE	Zn		Cd		Pb		Ni		B	
NO.	U	D	U	D	U	D	U	D	U	D
1	100	96	ND	ND	150	140	69	42	5.6	ND
2	58	62	0.6	ND	95	130	35	71	3.3	2
3	83	87	ND	ND	150	210	51	97	17	18
4	120	120	ND	ND	170	150	48	55	17	14
5	120	100	ND	ND	150	160	55	48	14	14
6	95	63	ND	ND	200	250	85	110	19	21
7	63	60	ND	ND	250	170	110	82	21	16
8	160	250	0.5	0.9	230	340	74	110	17	24
9	160	160	ND	ND	240	310	130	91	13	15
10	220	120	0.6	ND	320	220	120	76	19	18
11	110	87	ND	ND	230	230	120	110	20	19
12	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
13	59	140	ND	0.6	220	140	39	47	42	16
14	290	260	ND	ND	250	200	37	65	54	25
15	220	310	ND	0.7	150	170	23	16	20	23
16	65	57	ND	ND	130	160	29	33	14	26
17	57	99	ND	ND	160	160	33	23	26	34
18	160	250	0.5	1	150	250	35	100	22	18
19	120	210	0.8	0.8	210	300	61	48	32	30
20	94	84	ND	ND	150	140	44	54	20	14
21	84	99	ND	ND	140	180	54	41	14	16
22	99	87	ND	ND	180	140	41	41	16	13
23	87	80	ND	ND	140	120	41	32	13	9.6
24	2800	990	ND	1.5	950	350	310	82	150	56
25	990	570	1.5	ND	350	300	82	98	56	31
26	570	370	ND	ND	300	350	98	130	31	38
27	320	210	0.8	ND	330	210	130	39	23	14
28	130	110	6.5	5.5	360	260	53	47	180	140
29	23	110	ND	ND	78	340	27	50	12	100
30	270	50	2.8	ND	480	130	54	53	42	13
31	53	44	ND	ND	140	200	69	100	21	19
32	ND	ND	ND	ND	100	130	89	82	ND	ND
33	160	120	ND	ND	280	180	120	46	30	17
34	46	29	ND	ND	180	110	79	35	14	6.9
35	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
36	45	73	ND	ND	130	140	18	12	10	14
37	73	84	ND	ND	140	210	12	43	14	16
38	78	120	ND	ND	270	280	56	29	23	30
39	90	150	ND	ND	240	250	51	36	22	25
40	58	27	ND	ND	170	170	48	100	18	11
41	130	140	13.2	24	240	222	50	52	45	56
42	390	1300	1.4	4.4	340	540	32	53	25	77
43	36	36	ND	ND	90	110	42	84	1	1.7
44	88	47	ND	ND	140	95	49	26	10	ND
45	100	150	ND	ND	200	210	140	130	24	20
46	16	17	ND	ND	100	110	72	63	ND	9.9
47	89	50	0.7	ND	150	130	57	78	19	7.5

TABLE 7.10 continued

SITE	Cr		V		Cu		Ag		As	
NO.	U	D	U	D	U	D	U	D	U	D
1	7	4	12	8	10	8	ND	ND	2.4	1.6
2	3	4	8	8	ND	2	ND	ND	3.1	1.9
3	10	11	12	15	15	20	ND	ND	1.8	1.7
4	9	10	13	13	25	26	ND	ND	1.7	1.6
5	10	11	13	13	26	34	ND	ND	1.6	1.7
6	9	9	13	12	19	16	ND	ND	3.4	2.8
7	9	7	12	12	16	12	ND	1	2.8	3.6
8	13	19	14	19	29	53	ND	1.1	2	3
9	11	11	12	14	32	26	1	ND	1.7	2.1
10	15	13	18	12	48	29	ND	2.7	2.6	1.8
11	10	10	14	13	28	35	ND	1.4	3.5	3.2
12	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
13	24	16	19	11	19	28	ND	31.6	2.6	3.2
14	86	29	38	26	24	14	ND	ND	2.9	2.6
15	10	12	17	17	11	10	ND	ND	2.9	2.5
16	7	13	11	14	ND	6	ND	ND	2.5	2.5
17	13	15	14	15	6	8	ND	ND	2.5	3.3
18	16	19	12	25	27	40	ND	ND	3.1	4.9
19	13	21	26	25	43	103	ND	ND	2.9	2.5
20	11	6	16	11	10	5	ND	ND	1.9	1.4
21	6	14	11	15	5	19	ND	ND	1.4	1.7
22	14	9	15	13	19	11	ND	ND	1.7	1.3
23	9	7	13	11	11	8	ND	2	1.3	1.5
24	46	19	26	20	1600	340	ND	ND	1.1	2.5
25	19	14	20	15	340	160	ND	ND	2.5	3.4
26	14	19	15	21	160	120	ND	ND	3.4	2
27	8	12	11	9	31	60	ND	ND	2.4	1.7
28	34	22	56	45	12	10	ND	ND	15	14
29	3	17	7	34	7	12	ND	ND	2	10
30	12	8	17	7	25	1	ND	ND	2.7	1.1
31	5	7	9	8	1	4	ND	ND	1.3	1.7
32	ND	1	ND	1	ND	ND	ND	ND	1.3	1.5
33	9	6	19	11	25	20	ND	ND	2	1.5
34	6	2	10	7	15	10	ND	ND	2.4	1.9
35	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
36	6	7	6	10	8	13	ND	ND	1.7	2.6
37	7	7	10	10	13	17	ND	ND	2.6	2.5
38	10	12	15	18	18	24	ND	ND	2.8	3.5
39	8	14	12	17	19	29	ND	ND	2.8	3.5
40	10	5	11	8	25	10	ND	ND	3.2	2.4
41	25	26	22	27	64	26	9.9	ND	2.1	2.6
42	64	81	24	46	81	270	1.1	ND	2.2	2.2
43	3	3	1	ND	4	6	2.2	ND	1.6	1.4
44	9	3	8	1	5	3	ND	ND	ND	ND
45	8	13	9	15	17	23	ND	ND	1.4	2.3
46	6	6	5	7	ND	ND	ND	7.3	ND	1.1
47	9	6	12	8	11	5	ND	ND	1.7	1.3

ND = not detected; NS = not sampled; LT = less than

German rivers. For zinc, the average values from the present study were lower, with only the highest recorded concentrations similar to the high levels in the German rivers. The German lead values were also generally higher.

Overall, there was little difference between upstream and downstream metals concentrations; but as with BMWP scores there were considerable differences at some sites. Relationships between metals and upstream/downstream differences are investigated in Chapters 8 and 9 respectively.

7.3.2 Metals in algae

Cladophora was present at 32 of the sites monitored. Summary statistics for metals in algae are given in Table 7.11, with full results in Table 7.12. As for sediment samples, selenium and beryllium were not detected and therefore not considered further. The summary statistics have been calculated as described above for sediment samples.

In contrast to the results obtained for sediment samples, metals concentrations in *Cladophora* are higher than those quoted for other studies. Forstner and Wittmann (1981) have compiled a summary of observed metals concentrations in *Cladophora*, with the following concentration ranges (mg/kg) in the dry alga:

zinc	0.53 - 375
cadmium	0.05 - 22.9
copper	0.8 - 31.7
lead	5.2 - 347
nickel	11.9 - 23.8

For zinc and copper, the maximum concentrations above are exceeded by the mean values in Table 7.11, and for all metals except cadmium, the maximum values in Table 7.11 are higher than those reported for other studies. Although the possibility of contamination by fine sediment associated with the algae cannot be ruled out, the results indicate high levels of soluble metals in the sampled streams.

TABLE 7.11 SUMMARY OF METALS IN ALGAE SAMPLES

All concentrations in mg/kg dry algae

		MEAN	MEDIAN	STD.DEV	MIN	MAX	DF %
Zinc	U	379.9	280	316.6	37	1600	100
	D	399.5	345	363.5	47	2100	100
Cadmium	U	2.222	1.7	1.735	0.5	7.5	100
	D	2.184	1.5	1.715	0.7	7.5	100
Lead	U	193.7	130	185.5	LT 1	970	96.9
	D	178.2	120	154.0	LT 1	820	96.9
Nickel	U	13.19	8.5	13.34	LT 1	67	93.8
	D	16.16	10	23.28	LT 1	130	96.9
Boron	U	105.4	93	56.27	5.9	250	100
	D	104.1	97	41.87	15	200	100
Chromium	U	14.53	9.5	20.60	2	120	100
	D	14.91	10	25.54	2	150	100
Vanadium	U	15.25	13	9.834	2	36	100
	D	15.81	15	10.56	1	38	100
Copper	U	81.22	37.5	224.3	4	1300	100
	D	50.09	34.5	77.31	5	460	100
Silver	U	-	LT 1	-	LT 1	11	40.6
	D	-	LT 1	-	LT 1	9.4	46.9
Arsenic	U	2.075	1.9	1.560	LT 1	5.2	78.1
	D	2.275	2.0	1.374	LT 1	5.2	90.6

DF = Detection frequency; U = upstream; D = downstream

There was generally little difference between upstream and downstream metals concentrations; but as with BMWP scores and metals in sediments there were considerable differences at some sites. Relationships between metals, relationships with metals in sediments and upstream/downstream differences are investigated in Chapters 8 and 9.

TABLE 7.12 METAL CONCENTRATIONS IN ALGAE SAMPLES

All concentrations in mg/kg dry algae

SITE NO.	Zn U	Zn D	Cd U	Cd D	Pb U	Pb D	Ni U	Ni D	B U	B D
3	430	480	6.5	5	130	110	13	31	77	140
4	180	217	1.1	7.5	160	290	8	9	68	93
5	217	270	7.5	1.4	290	230	9	11	93	51
6	320	710	1.6	3.8	160	ND	15	26	54	110
7	710	450	3.8	2.4	ND	110	26	18	110	83
8	370	350	2	1.8	250	250	20	15	63	79
10	330	180	2	1.7	180	110	22	6	120	170
11	390	390	2.8	2.5	310	240	19	16	81	81
13	190	240	0.7	1	82	110	17	23	210	200
14	890	740	2.9	1.5	380	280	30	15	110	120
16	190	230	0.9	0.7	110	120	5	6	41	56
17	230	220	0.7	0.8	120	97	6	3	56	58
18	130	250	0.5	1.1	59	110	8	14	120	66
21	92	120	0.5	0.9	67	30	4	ND	26	86
24	430	340	1.9	1.1	220	96	8	5	59	90
25	340	480	1.1	1.6	96	120	5	9	90	93
26	480	760	1.6	3.6	120	120	9	15	93	94
30	520	460	3.5	3.2	390	160	6	3	110	100
31	510	500	4.4	4	240	350	10	9	73	100
32	37	47	1	1.1	29	39	2	3	5.9	15
34	160	120	1.1	1	130	130	2	5	95	130
36	220	240	1.9	1.7	220	230	5	7	88	91
37	240	170	1.7	1.4	230	150	7	1	91	110
38	190	420	1.4	3.3	130	320	1	8	140	100
39	160	170	1.4	1.1	110	150	ND	1	130	140
40	130	90	1.1	0.9	83	80	ND	3	210	170
42	1600	2100	5.6	7	970	820	28	130	49	35
43	660	500	0.9	1.1	96	75	67	46	250	180
44	430	470	1.7	1.5	100	65	33	35	170	140
45	240	360	1.2	0.9	120	150	6	16	160	140
46	150	140	2.8	1.8	45	110	12	16	190	91
47	990	570	3.3	1.5	570	450	19	12	140	120

continued on next page

TABLE 7.12 continued

All concentrations in mg/kg dry algae

SITE NO.	Cr U	Cr D	V U	V D	Cu U	Cu D	Ag U	Ag D	As U	As D
3	10	8	20	30	45	56	ND	3.5	3.9	4.1
4	9	12	12	36	37	33	1	ND	2.5	5.2
5	12	14	36	14	33	55	ND	1	5.2	2.7
6	11	3	16	30	44	17	1.4	6.6	1.6	2.5
7	3	13	30	20	17	37	6.6	7.2	2.5	2.4
8	15	10	20	17	70	70	1.1	1.6	2.3	2.1
10	9	6	17	8	64	59	2.2	ND	ND	ND
11	14	11	17	17	110	87	1.3	3.8	ND	ND
13	22	30	5	8	26	37	ND	ND	2.4	2.6
14	37	19	35	20	66	33	ND	ND	4.2	3.6
16	7	8	8	9	25	28	ND	ND	1.8	1.1
17	8	6	9	7	28	25	ND	ND	1.1	3
18	10	14	2	6	22	34	ND	ND	ND	2.3
21	7	2	7	5	150	10	ND	ND	1.6	ND
24	13	7	24	19	56	37	1.6	2.3	5.1	4.1
25	7	16	19	25	37	59	2.3	4.7	4.1	4.1
26	16	10	25	36	59	65	4.7	5.3	4.1	4.4
30	5	5	6	4	9	5	ND	ND	2.2	1.2
31	10	11	7	10	17	23	ND	ND	2.2	2.7
32	9	5	2	3	4	5	ND	ND	1.8	1.6
34	4	5	12	10	35	35	ND	ND	ND	1.7
36	8	9	24	23	42	47	ND	ND	1.4	1.6
37	9	3	23	18	47	24	ND	ND	1.6	1.3
38	3	5	17	38	17	33	ND	2	1.1	1.9
39	2	4	14	16	17	20	ND	ND	ND	1.4
40	3	11	12	9	13	13	ND	1.4	2.5	1.5
42	120	150	34	29	1300	460	11	9.4	4.2	4.4
43	21	15	2	1	57	57	2	1.2	ND	1.1
44	27	30	5	2	48	28	ND	ND	1.7	1.3
45	8	8	12	18	38	52	1.9	4.4	3.4	3.9
46	12	17	4	8	18	19	ND	ND	ND	1.2
47	14	10	12	10	48	40	2.6	1.2	1.9	1.8

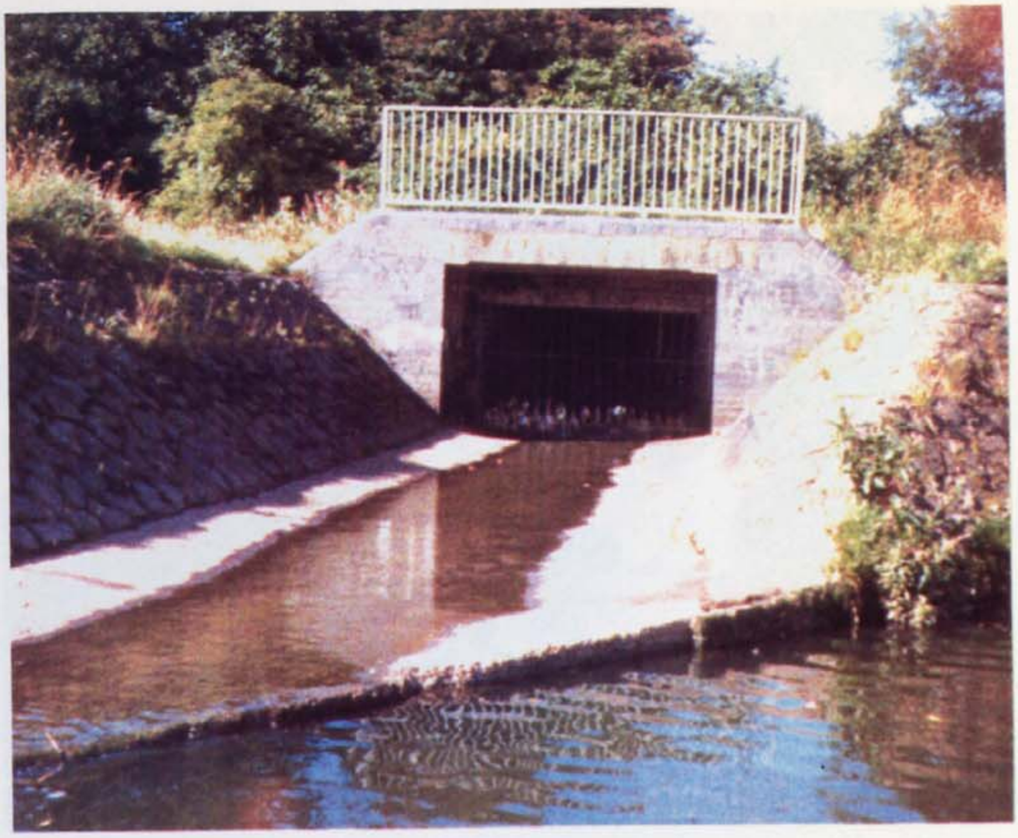
ND = not detected; U = upstream; D = downstream

(i) Outfall to artificial channel, Walsall

(ii) Outfall to artificial channel, Bromsgrove

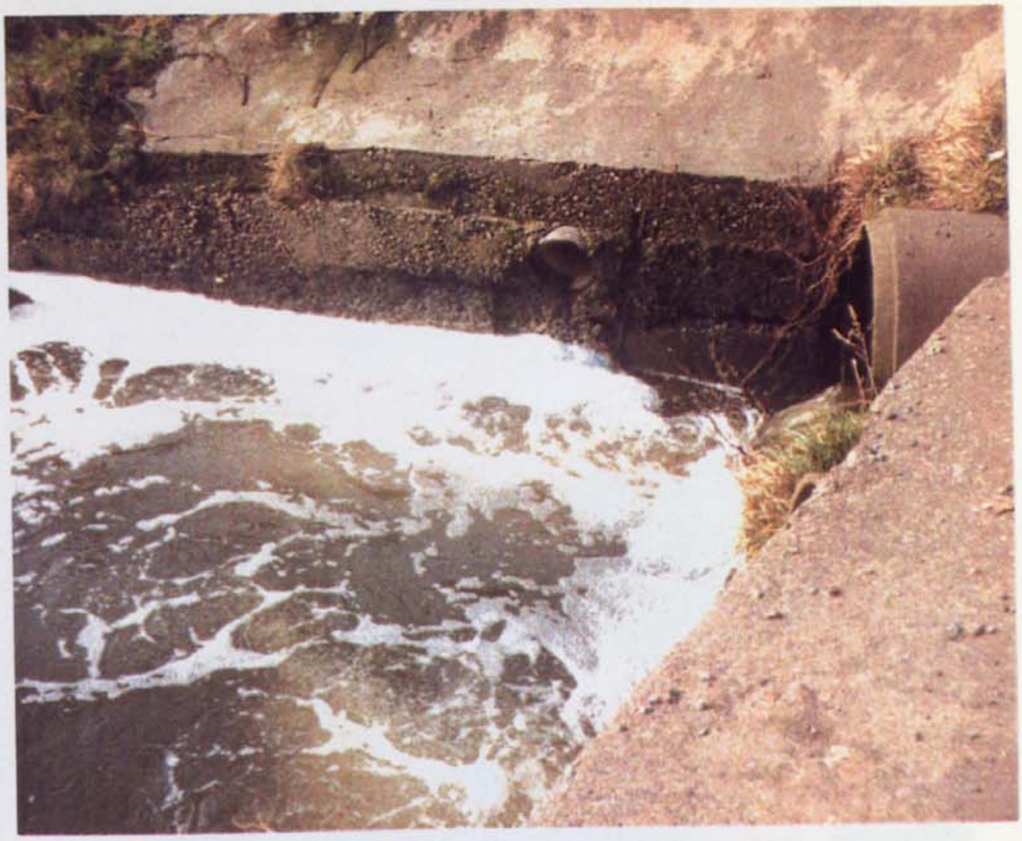
(iii) Site 15
Weed growth at outfall from airport

(iv) Site 15
Weed growth downstream of airport site



(v) Site 12
Outfall to R. Cole, Birmingham, showing dry weather sewage flow

(vi) Site 17
Outfall to Kingshurst Brook, Birmingham



(vii) Dry weather discharge from outfall in Walsall
(Not sampled because of culvert upstream)

(viii) Site 24
'Clean' dry weather discharge to Bourn Brook

(ix) Site 31
Outfall to R. Blithe, Staffs., showing
dry weather sewage flow

(x) Site 28
Twin outfalls to Whetstone Brook, Leics.
from an industrial estate and M1 motorway

(xi) Site 18
Outfall to R. Cole, Birmingham, with Chelmsley Wood estate in the background

(xii) Site 2
Outfall from construction site to R. Blythe, Solihull

(xiii) Site 35
Outfall to Battlefield Brook, Bromsgrove,
with sewered catchment in the background

(xiv) Site 30
Twin outfalls to R. Blithe, Staffs.

CHAPTER 8

RELATIONSHIPS BETWEEN MEASURED PARAMETERS

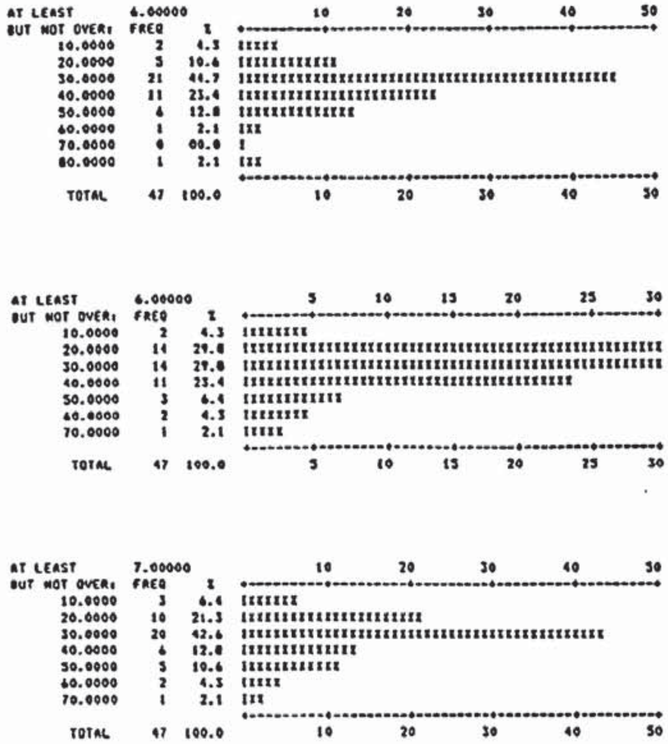
The purpose of this chapter is to investigate relationships between different parameters measured at the same sampling station. Knowledge of these relationships enables the relative utility of the various parameters to be assessed, and simplifies the analysis of station-to-station differences in Chapter 9. Relationships between different biological indices and between different metals are considered first, then relationships between biological parameters, metals concentrations, and catchment characteristics.

8.1 DATA TRANSFORMATIONS

Relationships were investigated using correlation analysis. The correlation coefficient, r, is meaningful only when the variables tested follow the bivariate normal distribution, so the biological indices and metals concentrations to be investigated were tested for normality. The nature and degree of departure from normality was assessed using the sample statistics for measuring skewness and kurtosis, and by inspection of histograms.

8.1.1 Biological indices

None of the biological indices used deviated seriously from normality, as shown in Figs. 8.1 to 8.4. BMWP scores were positively skewed, but not badly, and taxa counts, BMWP scores and comparative indices were all used in calculation of correlation coefficients without any transformation.

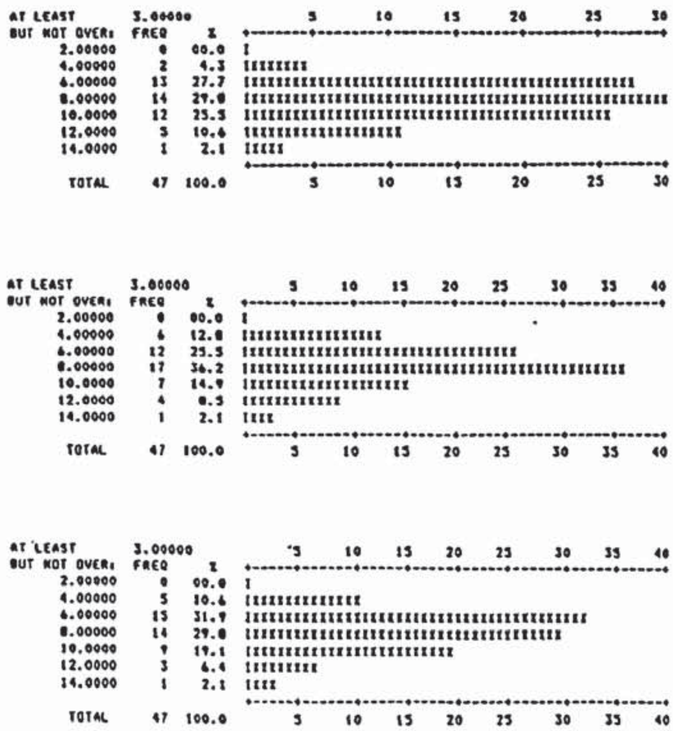


a) upstream stations

b) outfall stations

c) downstream stations

Fig. 8.1 BMWP Score Distribution



a) upstream stations

b) outfall stations

c) downstream stations

Fig. 8.2 Taxa Count Distribution

AT LEAST 10.0000

BUT NOT OVER:	FREQ	%
10.0000	1	2.1
20.0000	0	00.0
30.0000	5	10.6
40.0000	11	23.4
50.0000	13	27.7
60.0000	8	17.0
70.0000	5	10.6
80.0000	2	4.3
90.0000	1	2.1
100.000	1	2.1
TOTAL	47	100.0

a) upstream stations

AT LEAST 7.50000

BUT NOT OVER:	FREQ	%
10.0000	1	2.1
20.0000	5	10.6
30.0000	8	17.0
40.0000	9	19.1
50.0000	12	25.5
60.0000	3	6.4
70.0000	6	12.8
80.0000	2	4.3
90.0000	1	2.1
TOTAL	47	100.0

b) outfall stations

AT LEAST 10.5000

BUT NOT OVER:	FREQ	%
10.0000	0	00.0
20.0000	3	6.4
30.0000	6	12.8
40.0000	16	34.0
50.0000	9	19.1
60.0000	8	17.0
70.0000	1	2.1
80.0000	2	4.3
90.0000	1	2.1
100.000	1	2.1
TOTAL	47	100.0

c) downstream stations

Fig. 8.3 WBMWP Score Distribution

AT LEAST .283714

BUT NOT OVER:	FREQ	%
.100000	0	00.0
.200000	0	00.0
.300000	1	2.1
.400000	0	00.0
.500000	7	14.9
.600000	13	27.7
.700000	10	21.3
.800000	9	19.1
.900000	5	10.6
1.00000	2	4.3
TOTAL	47	100.0

a) Jaccard's Coefficient

AT LEAST 45.4545

BUT NOT OVER:	FREQ	%
10.0000	0	00.0
20.0000	0	00.0
30.0000	0	00.0
40.0000	0	00.0
50.0000	2	4.3
60.0000	5	10.6
70.0000	10	21.3
80.0000	14	29.8
90.0000	8	17.0
100.000	8	17.0
TOTAL	47	100.0

b) Kothe's Species Deficit

Fig. 8.4 Distribution of Comparative Indices

8.1.2 Metals concentrations

Metals concentrations in sediment and algae samples did not generally follow the normal distribution, as illustrated in Fig. 8.5 for copper in sediment samples and Fig. 8.6 for zinc in algae samples. Logs were taken of the metals concentrations and the distribution of the log values tested for normality using the third and fourth moment statistics, g_1 (skewness) and g_2 (kurtosis). These statistics are listed in Table 8.1.

TABLE 8.1 NORMALITY AND LOGNORMALITY OF METALS DATA

A - METALS IN SEDIMENT SAMPLES (n=45)

METAL	U/D	Conc. mg/kg		Log. conc.	
		g1	*g2*	*g1*	*g2*
Zinc	U	5.22	31	-0.77	8.5
	D	3.44	15	-1.10	8.4
Lead	U	3.36	17	0.76	4.4
	D	1.44	5.8	0.40	2.6
Nickel	U	2.82	14	0.12	3.7
	D	0.45	2.2	-0.60	3.2
Boron	U	3.54	16	-0.99	5.5
	D	2.77	11	-0.95	4.4
Chromium	U	3.13	13	0.19	4.9
	D	4.01	23	-0.39	4.5
Vanadium	U	2.37	11	-1.72	8.4
	D	1.53	5.9	-1.59	6.5
Copper	U	6.02	39	0.32	4.7
	D	3.28	14	0.04	3.3
Arsenic	U	5.01	31	1.43	8.7
	D	3.65	18	1.38	6.0

B - METALS IN ALGAE SAMPLES (n=32)

METAL	U/D	Conc. mg/kg $g1$	Conc. mg/kg $g2$	Log. conc. $g1$	Log. conc. $g2$
Zinc	U	2.13	8.2	0.16	3.5
	D	3.30	16	0.09	3.7
Cadmium	U	1.57	4.8	0.27	2.4
	D	1.79	5.6	0.74	2.6
Lead	U	2.62	11	-2.26	11
	D	2.48	11	-2.49	12
Nickel	U	2.23	9.3	-0.45	2.8
	D	3.86	19	-0.19	3.1
Boron	U	0.73	3.1	-1.75	7.9
	D	0.26	2.9	-1.55	6.7
Chromium	U	4.41	23	0.64	4.8
	D	4.84	26	0.99	5.7
Vanadium	U	0.56	2.4	-0.74	2.8
	D	0.60	2.3	-0.91	3.6
Copper	U	5.25	29	1.14	7.3
	D	4.85	26	0.21	5.4
Arsenic	U	0.32	2.2	0.13	2.0
	D	0.31	2.3	0.12	1.7

For a normal distribution, $g_1 = 0$ and $g_2 = 3$

In the calculation of statistics for untransformed data, detection limit values were given a value of zero. Where the data to be transformed included zeros, however, the detection limit value of 1.0 was substituted to avoid errors caused by attempting to calculate the log of zero. For most metals there were very few detection limit values, and their presence had negligible effect, but for cadmium in sediments there were many such values and these data were not used.

For most metals, the lognormal distribution described the data well, as illustrated in Figs. 8.7 and 8.8. Exceptions to this were boron, vanadium and arsenic in sediments, lead in algae and boron in algae. As these non-lognormal distributions were in upstream/downstream pairs, it was considered likely that the underlying distribution was not lognormal, and a distribution which better described the data was sought by taking square roots of the concentrations.

AT LEAST 0.00000

BUT NOT OVER:	FREQ	%
100.000	42	93.3
200.000	1	2.2
300.000	0	00.0
400.000	1	2.2
500.000	0	00.0
600.000	0	00.0
700.000	0	00.0
800.000	0	00.0
900.000	0	00.0
1000.00	0	00.0
1100.00	0	00.0
1200.00	0	00.0
1300.00	0	00.0
1400.00	0	00.0
1500.00	0	00.0
1600.00	1	2.2
TOTAL	45	100.0

a) Upstream Samples

AT LEAST 0.00000

BUT NOT OVER:	FREQ	%
50.0000	38	84.4
100.000	2	4.4
150.000	2	4.4
200.000	1	2.2
250.000	0	00.0
300.000	1	2.2
350.000	1	2.2
TOTAL	45	100.0

b) Downstream Samples

Fig. 8.5 Distribution of Copper in Sediment Samples

AT LEAST 37.0000

BUT NOT OVER:	FREQ	%
100.000	2	6.2
200.000	9	28.1
300.000	5	15.6
400.000	5	15.6
500.000	4	12.5
600.000	2	6.2
700.000	1	3.1
800.000	1	3.1
900.000	1	3.1
1000.00	1	3.1
1100.00	0	00.0
1200.00	0	00.0
1300.00	0	00.0
1400.00	0	00.0
1500.00	0	00.0
1600.00	1	3.1
TOTAL	32	100.0

a) Upstream Samples

AT LEAST 47.0000

BUT NOT OVER:	FREQ	%
100.000	2	6.2
200.000	6	18.8
300.000	7	21.9
400.000	4	12.5
500.000	8	25.0
600.000	1	3.1
700.000	0	00.0
800.000	3	9.4
900.000	0	00.0
1000.00	0	00.0
1100.00	0	00.0
1200.00	0	00.0
1300.00	0	00.0
1400.00	0	00.0
1500.00	0	00.0
1600.00	0	00.0
1700.00	0	00.0
1800.00	0	00.0
1900.00	0	00.0
2000.00	0	00.0
2100.00	1	3.1
TOTAL	32	100.0

b) Downstream Samples

Fig. 8.6 Distribution of Zinc in Algae Samples

```
AT LEAST        0.00000          5      10     15     20     25      30     35     40
BUT NOT OVER:   FREQ      %   +------+-------+------+------+------+-------+------+------+
       1.00000     5   11.1   IIIIIIIIIIIIIII
       2.00000     5   11.1   IIIIIIIIIIIIIII
       3.00000    17   37.8   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       4.00000    13   28.9   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       5.00000     2    4.4   IIIIIII
       6.00000     2    4.4   IIIIIII
       7.00000     0   00.0   I
       8.00000     1    2.2   IIII
                              +------+-------+------+------+------+-------+------+------+
         TOTAL    45  100.0          5      10     15     20     25      30     35     40
```

a) Upstream Samples

```
AT LEAST        0.00000          5      10     15     20     25      30     35     40
BUT NOT OVER:   FREQ      %   +------+-------+------+------+------+-------+------+------+
       1.00000     4    8.9   IIIIIIIIIIII
       2.00000     6   13.3   IIIIIIIIIIIIIIIIII
       3.00000    17   37.8   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       4.00000    12   26.7   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       5.00000     3    6.7   IIIIIIIII
       6.00000     3    6.7   IIIIIIIII
                              +------+-------+------+------+------+-------+------+------+
         TOTAL    45  100.0          5      10     15     20     25      30     35     40
```

b) Downstream Samples

Fig. 8.7 Lognormal Distribution of Copper in Sediment Samples

```
AT LEAST        3.61092               10       20       30       40        50       60
BUT NOT OVER:   FREQ      %   +--------+--------+--------+--------+---------+--------+
       1.00000     0   00.0   I
       2.00000     0   00.0   I
       3.00000     0   00.0   I
       4.00000     1    3.1   IIII
       5.00000     3    9.4   IIIIIIIIII
       6.00000    17   53.1   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       7.00000    10   31.3   IIIIIIIIIIIIIIIIIIIIIIIIIIII
       8.00000     1    3.1   IIII
                              +--------+--------+--------+--------+---------+--------+
         TOTAL    32  100.0           10       20       30       40        50       60
```

a) Upstream Samples

```
AT LEAST        3.85015                 10         20         30         40         50
BUT NOT OVER:   FREQ      %   +----------+----------+----------+----------+----------+
       1.00000     0   00.0   I
       2.00000     0   00.0   I
       3.00000     0   00.0   I
       4.00000     1    3.1   IIII
       5.00000     4   12.5   IIIIIIIIIIIIII
       6.00000    14   43.7   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       7.00000    12   37.5   IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
       8.00000     1    3.1   IIII
                              +----------+----------+----------+----------+----------+
         TOTAL    32  100.0             10         20         30         40         50
```

b) Downstream Samples

Fig. 8.8 Lognormal Distribution of Zinc in Algae Samples

For boron and arsenic in sediments, this transformation did not improve the distribution, and as the log transformed data were closer to the normal distribution than the untransformed data, log transformations were used. Lead and boron in algae and vanadium in sediments were reasonably well described by the normal distribution when square roots were taken, and this transformation was used. The square root transformation was also used for vanadium in algae which was better represented by this distribution than by the lognormal distribution.

To summarise, log transformed data were used for metals concentrations except lead in algae, boron in algae, and vanadium; for which the square root transformation was used.

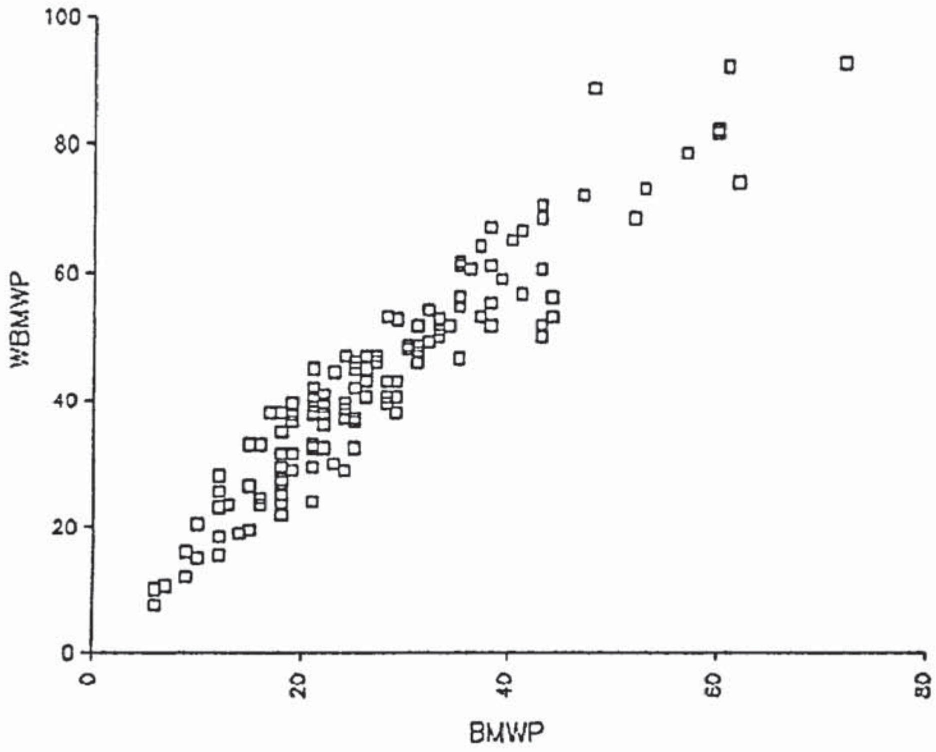
8.1.3 Other data and anova

In a few cases, variables other than biological indices and metals concentrations were used in correlation analysis. Any transformations performed on these are described as they occur in the text.

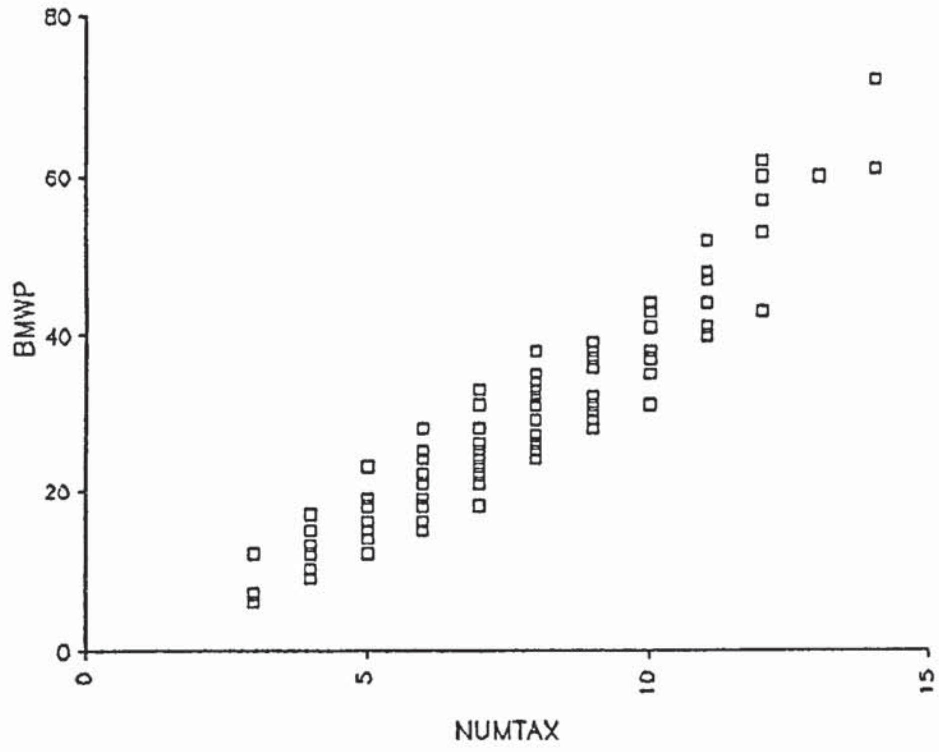
Analysis of variance (anova) was used to investigate the effects of some catchment characteristics on measured quality (section 8.6). The assumptions of anova are described in section 6.3, with tests to ensure that the assumptions are satisfied. The results of these tests are given when anova is used.

8.2 RELATIONSHIPS BETWEEN DIFFERENT BIOLOGICAL INDICES

As expected from the way in which scores are calculated, the agreement between taxa counts, BMWP scores and WBMWP scores was extremely high (Fig. 8.9), with values of r from 0.87 (WBMWP and taxa count) to 0.94 (BMWP and taxa count, $P < 0.001$).



a) BMWP and Weighted BMWP Scores



b) Taxa Counts and BMWP Scores

Fig. 8.9 Relationships Between Taxa Counts and BMWP Scores

Correlation between the two comparative indices, Jaccard's coefficient (J) and Kothe's Species Deficit (DM), was also very highly significant ($P<0.001$), as illustrated by Fig. 8.10. All points are below the line A-A, since if n% of taxa are lost at the downstream station, the similarity measured by Jaccard's coefficient can not be greater than (100-n)/100. Points which fall on the line A-A represent sites where upstream taxa were missing downstream without being replaced by different taxa. Points which fall on the line B-B are sites where none of the upstream taxa disappear, but additional taxa are present downstream. The other points represent sites where some taxa disappear and are replaced by others: the further the point from line A-A, the more replacement taxa downstream.

The application of the two comparative indices in this way was considered useful for investigating upstream/downstream differences and was employed as described in Chapter 9.

The relationship between BMWP scores and the two comparative indices was investigated using the BMWP score difference between the upstream and downstream stations. BMWP score differences are plotted against Kothe's Species Deficit and Jaccard's coefficient in Fig. 8.11, where negative BMWP differences indicate a drop in quality downstream of the outfall and positive differences indicate an improvement.

In Fig. 8.11 (a), there is correlation between negative BMWP differences and values of K, as disappearing taxa cease to contribute to the BMWP score. Comparison of K values with positive BMWP differences is meaningless as K takes no account of replacement taxa.

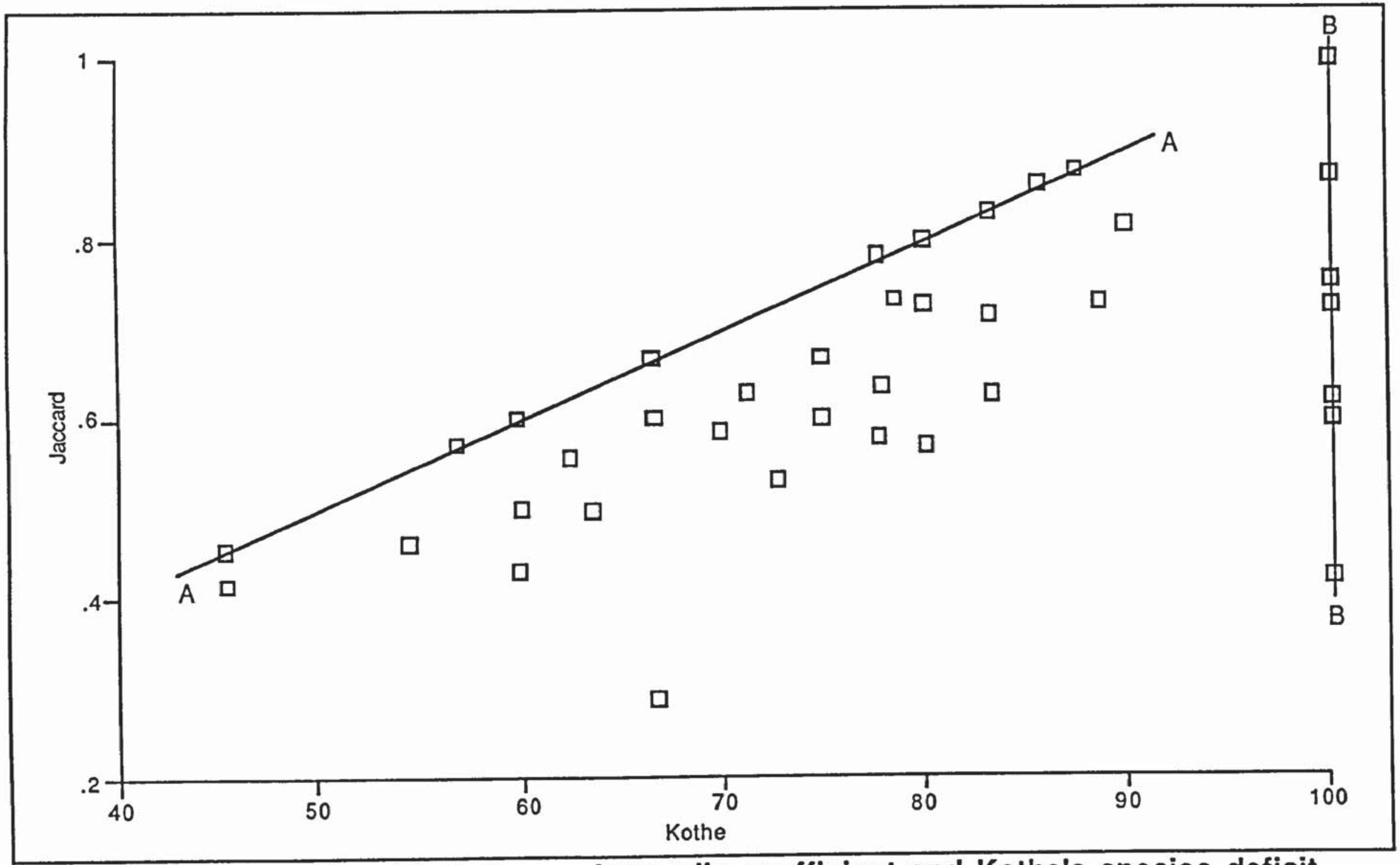
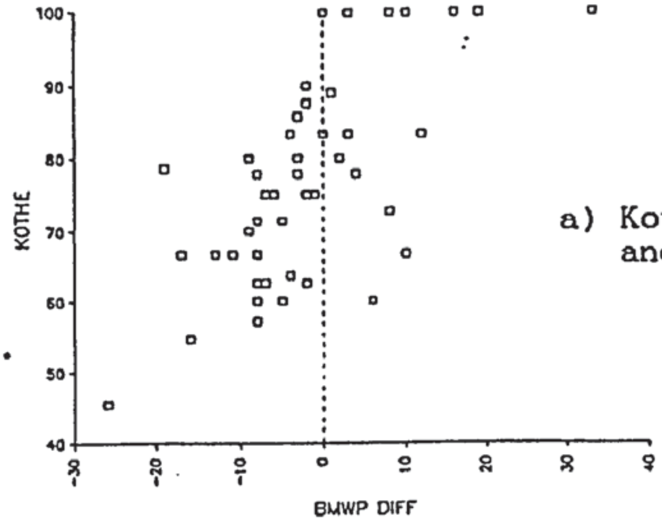
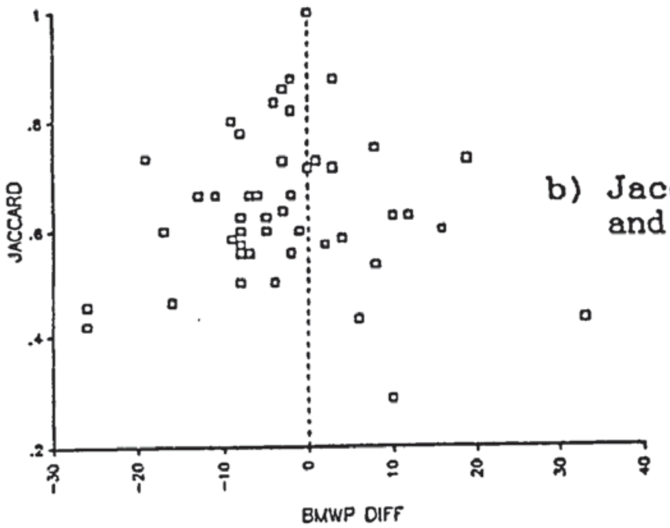


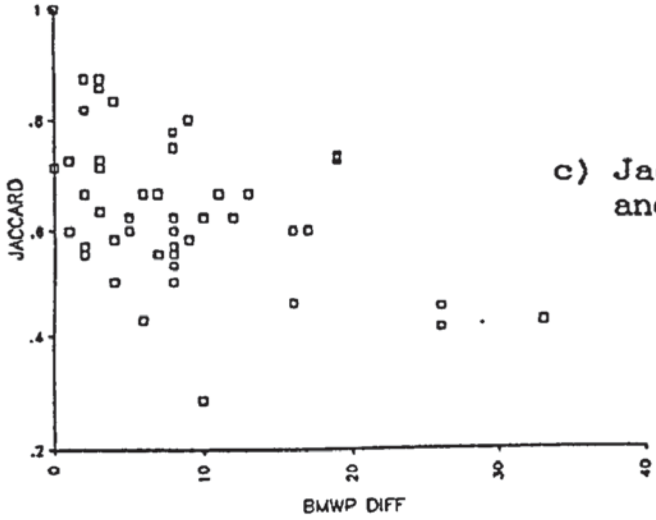
Fig 8.10 **Relationship between Jaccard's coefficient and Kothe's species deficit**



a) Kothe's Species Deficit
and BMWP Score Differences



b) Jaccard's Coefficient
and BMWP Score Differences



c) Jaccard's Coefficient
and Absolute Score Differences

Fig. 8.11 Relationships Between Comparative Indices and BMWP Scores

There is weak correlation between values of J and BMWP score differences, as shown by the inverted V shape of the plotted points in Fig. 8.11 (b). The strength of the correlation was measured by ignoring the direction of the score difference (Fig. 8.11 (c)) and calculating r, which was -0.4842 and highly significant.

Although there is close association between BMWP scores and the comparative indices used, the scatter of points in the graphs of Fig. 8.11 confirms that the two approaches reflect different upstream/downstream effects.

8.3 RELATIONSHIPS BETWEEN DIFFERENT METALS

8.3.1 Different metals in sediment samples

Using the median upstream and downstream concentrations for each metal, they were ranked in the following order of abundance:

1	Lead	(180,180)
2	Zinc	(95,99)
3	Nickel	(54,53)
4	Boron	(19,17)
5	Copper	(18,16)
6	Vanadium	(13,13)
7	Chromium	(10,11)
8	Arsenic	(2.2,2.4)
9	Silver)	(not
	Cadmium)	detected)

Lead, zinc and nickel were the most abundant metals, and were present in a typical sediment in the approximate ratio 4:2:1. Boron, vanadium, chromium and copper were less abundant, followed by arsenic which was present in small quantities.

The correlation matrix for different metals in upstream and in downstream sediment samples is shown in Table 8.2. The values of r were all calculated using transformed data as described in section 8.2.1.

Most metals in sediments correlated well ($P<0.001$). The only exceptions were nickel, which was associated with lead but no other metals; and arsenic, for which correlation with other metals was usually significant but weak. The agreement between values of r calculated using upstream data and those calculated using downstream data was good.

The analysis indicated that high concentrations of zinc, lead, boron, chromium, vanadium and copper were found simultaneously in sediment samples. Nickel and arsenic concentrations were not generally related to other metals. The difference in behaviour of zinc and arsenic was attributed to either a difference in chemistry of these metals or to variation in their supply to different sites.

8.3.2 Different metals in algae samples

Using the median upstream and downstream concentrations for each metal, they were ranked in the following order of abundance:

1	Zinc	(280,345)
2	Lead	(130,120)
3	Boron	(93,97)
4	Copper	(37.5,34.5)
5	Vanadium	(13,15)
6	Chromium	(9.5,10)
7	Nickel	(8.5,10)
8	Arsenic	(1.9,2.0)
9	Cadmium	(1.7,1.5)
10	Silver	(ND,ND)

The dominant metals in the algae samples were lead and zinc, as in sediment samples, followed by boron and copper which were not as abundant in sediments. Nickel, which was abundant in sediments, did not appear in algae to the same extent. The difference in the order of abundance of the metals in the two sample types was attributed to differential bioavailability of metals and to preferential uptake by algae.

The correlation matrix for different metals in upstream and in downstream algae samples is shown in Table 8.3. The values of r were all calculated using data transformed as described in section 8.2.1.

Metals in algae samples did not show such good correlation as metals in sediment samples, probably because of differences in bioavailability between metals and preferential uptake of some elements by the plants. At the 5% level there was no significant correlation between boron and any other metals, or between chromium and cadmium, nickel and lead, arsenic and lead, vanadium and nickel, or vanadium and chromium. Highly significant correlation between metals was less common than in sediment samples, with the strongest association between nickel and chromium.

TABLE 8.2 CORRELATION BETWEEN DIFFERENT METALS IN SEDIMENT SAMPLES

Values of r in pairs for upstream (upper value) and downstream (lower value)

n=45, P=0.001 r=.4583
P=0.05 r=.2543

Zn	1.0000 1.0000							
Pb	.7624 .6759	1.0000 1.0000						
Ni	.2804 .0025	.5505 .3358	1.0000 1.0000					
B	.7232 .6197	.7587 .7253	.1714 .0288	1.0000 1.0000				
Cr	.7470 .8203	.7054 .7533	.0822 .0494	.7676 .8339	1.0000 1.0000			
V	.6867 .6769	.6669 .7586	.0698 .0070	.8585 .8596	.8667 .8699	1.0000 1.0000		
Cu	.8029 .8148	.8021 .7747	.4086 .1817	.6774 .5713	.6794 .6874	.5560 .5805	1.0000 1.0000	
As	.1929 .3073	.2847 .5514	-.1379 .0097	.4560 .6363	.3163 .4774	.5736 .6688	.1566 .3297	1.0000 1.0000
	Zn	Pb	Ni	B	Cr	V	Cu	As

TABLE 8.3 CORRELATION BETWEEN DIFFERENT METALS IN ALGAE SAMPLES

Values of r in pairs for upstream (upper value) and downstream (lower value)

n=32, P=0.001 r=.5415
P=0.05 r=.3061

Zn	1.0000 1.0000							
Cd	.5856 .5723	1.0000 1.0000						
Pb	.5953 .4618	.5120 .4228	1.0000 1.0000					
Ni	.6884 .6963	.3969 .4339	.2669 .2830	1.0000 1.0000				
Cr	.5136 .5103	.2993 .2557	.5571 .5373	.7288 .7584	1.0000 1.0000			
Cu	.5449 .6169	.2516 .3108	.5945 .5800	.4920 .6541	.6686 .6323	1.0000 1.0000		
B	.2209 .0466	-.0375 -.1468	-.1171 -.1936	.1798 .0528	-.0466 -.0626	-.1193 .0105	1.0000 1.0000	
V	.4734 .4331	.5278 .6178	.4195 .3384	.1120 .1938	.1373 .0462	.4584 .4532	-.1628 -.0814	1.0000 1.0000
As	.3648 .4630	.4101 .4111	.2833 .2774	.1635 .3765	.3411 .3544	.2210 .4014	-.1735 -.1454	.5533 .6319
	Zn	Cd	Pb	Ni	Cr	Cu	B	V

8.4 RELATIONSHIPS BETWEEN METALS IN SEDIMENTS AND METALS IN ALGAE

Median concentrations of metals in upstream and downstream sediment and algae samples are compared in Table 8.4. Chromium, vanadium, and arsenic concentrations were similar in the two sample types, although differences in moisture content between sediments and algae probably meant that this was not the case in fresh samples. For other metals, there were large differences in concentrations - the most significant being for zinc, boron and nickel. Zinc and boron concentrations were much higher in algae than in sediments, whereas nickel concentrations were lower. There were also differences in levels of copper, cadmium, and lead.

TABLE 8.4 COMPARISON OF METALS CONCENTRATIONS IN UPSTREAM AND DOWNSTREAM SEDIMENTS AND ALGAE SAMPLES

	MEDIAN CONCENTRATIONS (mg/kg DRY SAMPLE)	
METAL	SEDIMENTS	ALGAE
Zinc	95,95	280,345
Lead	180,180	130,120
Nickel	54,53	8.5,10
Boron	19,17	93,97
Chromium	10,11	9.5,10
Vanadium	13,13	13,15
Copper	18,16	37.5,34.5
Arsenic	2.4,2.2	1.9,2.0
Cadmium	ND,ND	1.7,1.5

Correlation coefficients were calculated for each metal at each station to establish the degree of association between metal concentrations in algae and those in sediments. The calculated values of r are given in Table 8.5.

TABLE 8.5 CORRELATION BETWEEN METALS IN SEDIMENTS AND METALS IN ALGAE

METAL	upstream samples	downstream samples
ZINC	0.5610	0.6155
LEAD	0.3103	0.3123
NICKEL	0.0367	0.2985
BORON	-0.0671	-0.1096
CHROMIUM	0.4404	0.3308
VANADIUM	0.6033	0.6397
COPPER	0.3706	0.6837
ARSENIC	-0.1234	0.2364

The relationships between metal concentrations in sediments and algae were significant for all metals except nickel, boron and arsenic. The closest associations were for zinc and vanadium.

Nickel, boron and arsenic generally appeared to behave differently from other metals: nickel and arsenic did not correlate with other metals in sediments, and boron did not correlate with other metals in algae. There was also a difference in the relative abundance of nickel and boron in sediments and algae. For these elements, bioavailability or preferential uptake by algae were apparently more significant than for other metals.

For metals other than nickel and boron, the concentration in algae reflected the concentration in the sediments.

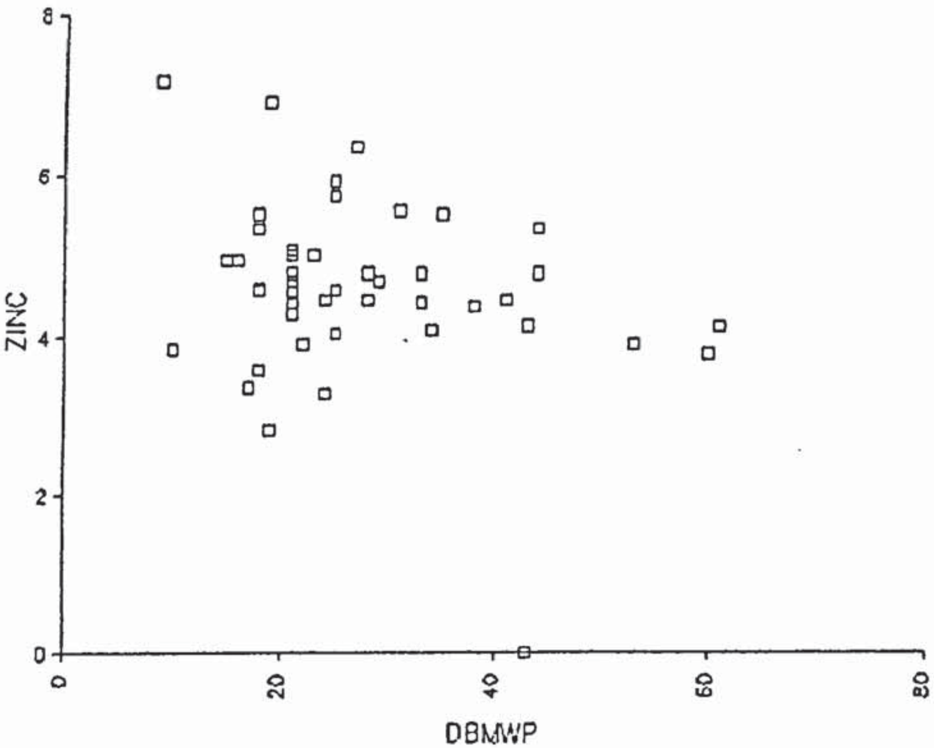
8.5 RELATIONSHIPS BETWEEN BIOLOGICAL INDICES AND METALS

To investigate the relationship between metals concentrations and biological quality, taxa counts and BMWP scores were compared with metals in sediments and metals in algae. Where association between biological parameters and metals was found, it generally applied to both taxa counts and BMWP scores; and for this reason only BMWP scores are discussed below.

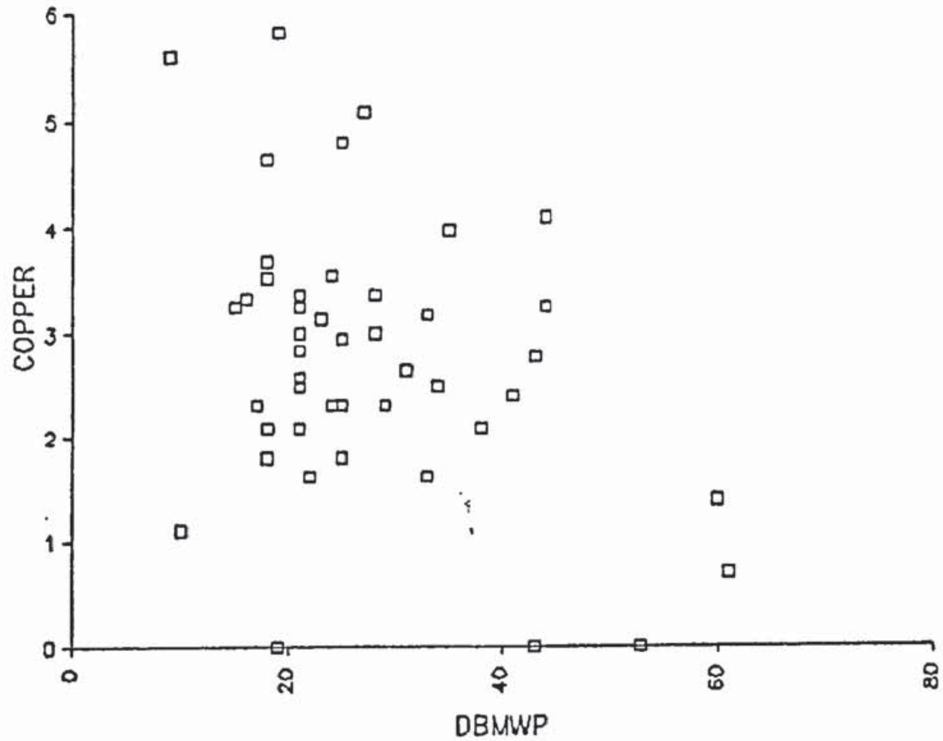
Correlation between metals in sediments and biological scores was poor. The only significant relationships found were for zinc and copper, and this was only for the samples taken at downstream stations. These relationships were weak, as illustrated in Fig. 8.12, although there was clearly a tendency for high metals concentrations to be associated with low BMWP scores. The pattern of the plotted points indicated that there may be a threshold level at which zinc and copper begin to affect BMWP scores, with an increasing proportion of taxa affected at higher concentrations.

In algae samples, correlation with BMWP scores was better, and significant relationships were found for chromium, copper and cadmium in both upstream and downstream samples (Fig. 8.13). The possible threshold effect observed in Fig. 8.12 was also apparent for chromium and copper in algae. BMWP scores decreased with increasing chromium and copper, but increased with cadmium concentrations (Fig. 8.13 (c)). The relationships were generally weak, however, and for most metals there was no significant correlation with biological scores, as shown in Fig. 8.13 (d) for zinc.

Metals concentrations in algae were expected to be more closely related to biological scores than metals in sediments, as the soluble forms of the elements available to algae are generally also available to macroinvertebrates. Although this was the case, the relationships between metals and biological scores for both sediment and algae samples were weak. This is probably because most of the metals concentrations are below the level at which they become toxic to macroinvertebrates and begin to affect biological indices.



a) Zinc



b) Copper

Fig. 8.12 Relationships Between Metals in Sediments and BMWP Scores

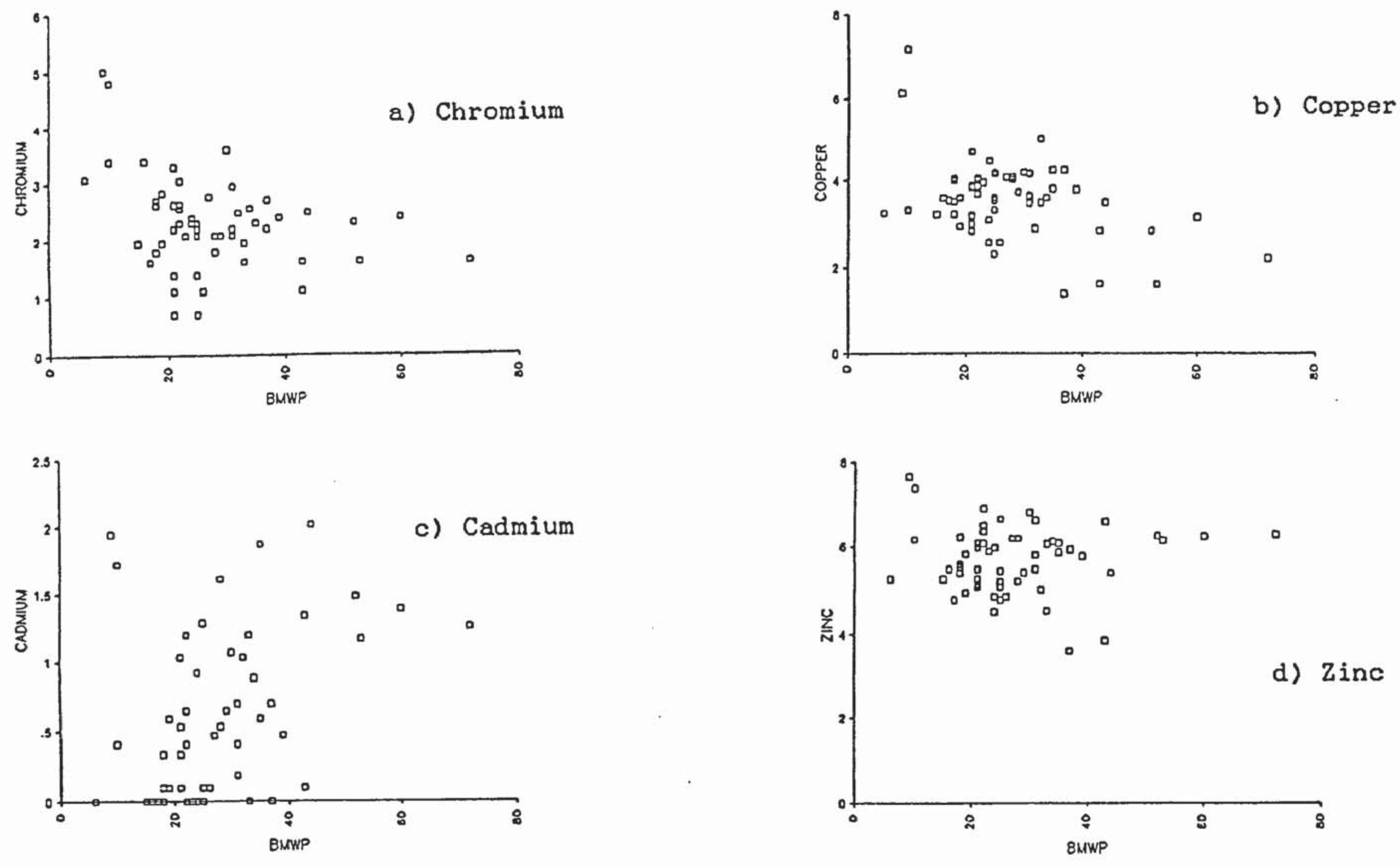


Fig. 8.13 Relationships Between Metals in Algae and BMWP Scores

8.6 THE EFFECT OF CATCHMENT CHARACTERISTICS ON BMWP SCORES AND METALS

The effects of upstream catchment area and degree of urbanisation upstream on measured quality were investigated using correlation analysis and anova. The relationship between NWC class and BMWP score was also investigated using anova.

The BMWP scores and metals concentrations used in the analysis were all from upstream stations, so that any localised effects of outfalls were minimised. It was assumed that BMWP was a general guide to biological quality, so taxa counts and WBMWP scores were not investigated.

8.6.1 BMWP scores and catchment characteristics

The relationship between BMWP scores and NWC class was investigated using single classification anova with two groups, for NWC classes 1B and 2. The groups were homoscedastic and approximately normal. The computed value of F was significant at the 1% level, indicating that NWC class has a highly significant effect on BMWP score, as might be expected for two parameters which indicate chemical and biological water quality respectively. The mean BMWP score at class 1B sites was 35.8, compared with a mean of 23.2 at class 2 sites. Any adverse effects on water quality due to outfalls that are detectable using BMWP scores are therefore likely to influence the NWC classification and could therefore affect the achievement of current water quality objectives.

BMWP scores were also analysed in relation to upstream catchment area and degree of urbanisation. It was anticipated that scores would generally fall with increasing values of both these parameters. The initial analysis was carried out using correlation, although there was some doubt over the accuracy of measurement of catchment characteristics (see section 5.6).

Catchment areas upstream of discharge points were skewed, and a square root transformation was applied to normalise the distribution. The value of r computed for BMWP scores and degree of urbanisation was -0.3344, which is significant at the 5% level. With upstream catchment area, however, the value of r was only -0.1865, which is not significant.

The reason for this unexpected result was probably that the small upstream areas were also highly urbanised, and the effects of urbanisation masked the effects of upstream area. This was confirmed by carrying out two anovas, one for each catchment characteristic. Degree of urbanisation was grouped into four classes:

rural	urbanisation < 25%
mostly rural	25% < urbanisation < 50%
mostly urban	50% < urbanisation < 75%
urban	urbanisation > 75%

Upstream catchment area (ha) was grouped into six classes:

A	area < 1000
B	1000 < area < 2000
C	2000 < area < 3000
D	3000 < area < 4000
E	4000 < area < 5000
F	area > 5000

For both tests the data in the different classes were homoscedastic and normal. The results of the anova for each classification were significant at the 5% level, proving that both upstream catchment area and degree of urbanisation influenced BMWP score. The mean BMWP scores in each urbanisation class were as follows:

rural	37.1
mostly rural	25.9
mostly urban	27.9
urban	23.7

The scores were highest for rural catchments and lowest for urban ones, but in mixed catchments there was little difference.

The mean BMWP scores in each upstream area class were:

A	26.7
B	36.3
C	36.0
D	24.3
E	27.0
F	18.8

As suspected, although the general trend was for scores to decrease as catchment area increased, the smallest catchments (class A) did not have the highest scores. Many of the sites with small upstream areas were urban tributaries of rivers, and the low mean BMWP score probably reflects the degree of urbanisation rather than the catchment area.

8.6.2 Metals concentrations and catchment characteristics

The relationships between upstream catchment area, degree of urbanisation and metals concentrations in sediments and algae were also investigated using correlation analysis and anova.

Correlation analysis was carried out using transformed metals data as described in section 8.2.1, and the square root transformation of upstream catchment area. The degree of urbanisation upstream was normally distributed. The values of r obtained are given in Table 8.6.

TABLE 8.6 CORRELATION BETWEEN CATCHMENT CHARACTERISTICS AND METALS IN SEDIMENTS AND ALGAE

		DEGREE OF URBANISATION UPSTREAM	UPSTREAM CATCHMENT AREA
ZINC	algae	0.3070	-0.2418
	sediments	**0.5473**	-0.0469
CADMIUM	algae	-0.1699	**-0.4270**
LEAD	algae	0.2301	**-0.3741**
	sediments	**0.3482**	-0.0976
NICKEL	algae	0.2363	0.0493
	sediments	-0.0004	-0.0437
CHROMIUM	algae	**0.4580**	-0.1217
	sediments	**0.6104**	-0.1176
COPPER	algae	**0.4700**	-0.2246
	sediments	**0.3778**	-0.0888
BORON	algae	0.0122	**0.3940**
	sediments	**0.3827**	-0.0666
VANADIUM	algae	0.1381	**-0.3846**
	sediments	**0.4186**	-0.2274
ARSENIC	algae	0.2483	**-0.3724**
	sediments	0.0490	-0.1139

Values of r significant at the 5% level in bold type

The metals in sediments which increased with urbanisation were zinc, lead, chromium, copper, boron and vanadium. There was no evidence of a relationship between urbanisation and nickel or arsenic. This corresponds with the elements quoted by Forstner and Wittmann (1981) as having high relative pollution potentials. The only metals in algae which increased with urbanisation were zinc, chromium and copper. The correlations were generally weak.

Correlation between metals and upstream catchment area was very poor. Most of the values of r were negative, an unexpected result which might be explained by the urban tributaries effect described in section 8.6.1.

Anova was carried out for copper in algae samples and degree of urbanisation upstream. Copper correlated well with degree of urbanisation and with other metals in algae samples. Although the means of the groups varied between 26.8 mg/kg Cu (rural catchments) and 199 mg/kg Cu (urban catchments), the variation within groups was also large, and the F ratio calculated was not significant, indicating that the degree of urbanisation had no effect on metals levels.

Inspection of the scatter plot of copper and degree of urbanisation revealed that the high correlation was due largely to extreme values in the data, which boosted the value of r but would not increase the value of F in an anova.

It was generally concluded that although there is a tendency for most metals concentrations to increase with degree of urbanisation upstream, the large variation in the levels of metals at all types of site was too great for the urbanisation effect to be clear.

8.7 SUMMARY

BMWP scores, taxa counts and WBMWP scores all showed very close association. Correlation between comparative indices and differences detected in BMWP scores was significant but weak, suggesting that the two revealed different effects. All the indices were used in the upstream/downstream analysis which follows in Chapter 9. A useful relationship between Jaccard's coefficient and Kothe's Species Deficit was found which was also used in the analysis of upstream/downstream effects.

Metals in sediments correlated well with each other, except for nickel and arsenic, but correlation between metals was not as good in algae samples. This was explained by differences in bioavailability and preferential uptake of some elements by algae, which was also suggested as an explanation for the difference in the relative abundance of metals in sediments and algae. Concentrations of metals in algae generally reflected concentrations in sediments, but this was not true for nickel, boron or arsenic.

Correlation between metals in sediments and BMWP scores was poor, and generally insignificant. Correlation between metals in algae and BMWP scores was better, presumably because of common availability, but was only significant for chromium, copper and cadmium. It was assumed that metals concentrations were generally not high enough to affect BMWP scores.

BMWP scores were significantly higher at NWC class 1B sites than at class 2 sites. The relationship this implies may be important where adverse effects on BMWP scores due to outfalls are detected, as it indicates that the presence of outfalls can prevent water quality objectives from being achieved.

It was demonstrated that BMWP scores are affected by the degree of urbanisation upstream : streams in rural catchments had significantly higher BMWP scores than streams in urban catchments. BMWP scores were also generally higher at points where the upstream catchment area was small, although the smallest catchments studied were urban and this masked the effect. Most metal concentrations also increased with the degree of urbanisation but large variation in concentrations reduced the significance of this effect.

CHAPTER 9

UPSTREAM - DOWNSTREAM DIFFERENCES

In this chapter the differences in biological quality and metals concentrations between sampling stations at the same site are investigated. For biological scores this involves three stations at each site, and for metals concentrations two stations at each site.

Differences are first described and their significance tested using the whole data set of 47 sites. This is carried out for biological parameters and metals separately, and then the two are compared. The data set is then divided into categories based on catchment characteristics, so that the effects of different characteristics can be assessed independently.

9.1 BIOLOGICAL DIFFERENCES

9.1.1 Taxa counts and BMWP scores

Taxa counts, BMWP scores, and WBMWP scores at the same station showed good agreement (section 8.2). Station-to-station differences in one were generally reflected by corresponding increases or decreases in the others.

There was no consistent pattern to the indices at different stations. In some cases there was a considerable decrease in all three, indicating a fall in quality downstream of the outfall, but in other cases all three indices increased, indicating an improvement in water quality downstream of the outfall.

Small changes of 10 or less in BMWP scores were treated with caution as they could have been the result of a single organism appearing or disappearing, and well within normal sampling error. Differences due to sampling error are as likely to be positive as negative, however, and a large number of small negative or positive differences would therefore be significant.

Upstream-outfall-downstream patterns

Because biological samples were taken at three stations, there were several possible patterns of scores. For example, there might be a high BMWP score upstream of the outfall which falls at the outfall but recovers to just below its upstream value at the downstream station. This pattern can be represented by - + (-), where the first '-' is the drop in score at the outfall, the '+' is the recovery between the outfall and downstream stations, and the final '(-)' indicates that there is an overall drop between the upstream and downstream stations.

The possible patterns using this notation are:

Pattern 1	- - (-)	
Pattern 2	- - (+)	*
Pattern 3	- + (-)	
Pattern 4	- + (+)	
Pattern 5	+ - (-)	
Pattern 6	+ - (+)	
Pattern 7	+ + (-)	*
Pattern 8	+ + (+)	

where * indicates that the pattern could not occur as two successive +'s can not add up to an overall (-) and vice versa.

Odd numbered patterns indicate that downstream quality is lower than upstream quality, and even numbered patterns indicate that downstream quality is higher than upstream quality.

The 47 sites were categorised according to the pattern their biological indices followed. Where the direction of change between stations was not the same for all the indices, the site was not classified. The results were as follows:

Pattern 1	7 sites
Pattern 3	7 sites
Pattern 4	5 sites
Pattern 5	7 sites
Pattern 6	2 sites
Pattern 8	5 sites
Unclassified	14 sites
	========
TOTAL	47 sites

Ignoring the unclassified sites, this distribution was compared with the expected distribution which could occur by chance. A Chi-square test showed that there was no significant difference between the two. There was therefore no consistent pattern caused by the presence of the outfall.

The odd and even numbered patterns were then considered. This enabled some of the previously unclassified sites to be included, with the following result:

Odd (overall decrease in quality)	28 sites
Even (overall increase in quality)	14 sites
Unclassified	5 sites

Twice as many sites had reduced indices downstream of the outfall than had increased indices. This distribution was also tested using a Chi-square test and found to be significant at the 5% level. The observed difference was unlikely to occur by chance and the presence of the outfall appeared to have a detrimental effect on water quality.

The investigation of patterns does not take into account the magnitude of the observed differences in scores. This can be considered by anova or t-tests for paired comparisons.

<u>The significance of upstream-outfall-downstream differences</u>

Upstream, outfall and downstream taxa counts, BMWP and WBMWP scores were further investigated with t-tests for paired comparisons. Each pair of indices compared was first tested for homoscedasticity with an F-test, and all variances were shown to be equal with 95% confidence. The calculated values of t are given in Table 9.1.

TABLE 9.1 VALUES OF t FOR PAIRED COMPARISON OF BIOLOGICAL SCORES AT DIFFERENT SAMPLING STATIONS

A. UPSTREAM vs DOWNSTREAM

Taxa counts	t=	-1.5590
BMWP scores	t=	-1.4515
WBMWP scores	t=	-2.1832

B. UPSTREAM vs OUTFALL

Taxa counts	t=	-1.4661
BMWP scores	t=	-1.6463
WBMWP scores	t=	-2.5622

C. OUTFALL vs DOWNSTREAM

Taxa counts	t=	-0.0121
BMWP scores	t=	+0.1262
WBMWP scores	t=	+0.2707

The values of t for upstream vs downstream and upstream vs outfall indices were all significant at the 10% level. The t values for WBMWP scores were significant at the 5% level. None of the t values for outfall vs downstream indices were significant.

The greater significance observed for decreases in WBMWP scores indicates that organism abundance was reduced at downstream sampling stations, an additional effect to the overall reduction in taxa and BMWP score. This is unlikely to be important at sites where there is no reduction in BMWP score, however, as a decrease in WBMWP score was usually accompanied by a decrease in BMWP score. There were a few sites where changes in quality were only apparent when abundance was considered, or where BMWP scores and WBMWP scores changed in opposite directions. This observation indicates that a more detailed biological examination, including identification to species level and use of more sophisticated indices, might reveal effects not detected by the approach adopted here.

To summarise; there is weak evidence of a decrease in taxa counts and BMWP scores downstream of outfalls, stronger evidence of a decrease in WBMWP scores, and no evidence of recovery 100m downstream. This supports the earlier pattern-count tests in which it was demonstrated that scores are as likely to go up as down between the outfall and downstream stations.

The significance of upstream-downstream differences is further investigated for sites with particular characteristics in section 9.5.

9.1.2 Comparative indices

Jaccard's coefficient and Kothe's Species Deficit were used together as described in section 8.2 and Fig. 8.10.

Two sites had identical taxa lists at the upstream and downstream stations, and six sites had new taxa at the downstream station without any of the upstream taxa disappearing. For these eight sites there was no evidence that quality was impaired by the outfall. These sites accounted for most of those at which the downstream BMWP score was higher than the upstream score, but not those at which taxa disappeared and were replaced by higher-scoring groups.

At ten sites, taxa disappeared downstream of the outfall with no replacement groups. These did not correspond with the sites where BMWP scores decreased the most, since in most cases a fall in BMWP score was the result of upstream taxa being replaced by lower-scoring groups. At these sites, particularly those with the lower coefficient values, the outfall was apparently damaging to water quality.

The remaining sites all had disappearing taxa and some replacement groups present downstream. For these sites it is more difficult to assess the impact of the outfall, but in general high impacts should be reflected by low coefficient values.

The different types of impact identified by the joint use of comparative indices are related to catchment characteristics in section 9.5.

9.2 DIFFERENCES IN METAL CONCENTRATIONS

Initial examination of metals results for sediment and algae samples did not reveal any overall tendency for an increase in concentrations downstream of the outfalls. Differences are summarised in Table 9.2 for each element in sediment and algae samples.

Metals in sediment deposits may have entered the river with the sediment, or the metals may have been adsorbed from solution. Metals in algae can only have been incorporated in solution from the water, although the sediment may have acted as a store. Sediments and algae would therefore be expected to reveal different types of pollution, although a severely contaminated discharge would probably be detected in both; provided that contamination upstream was not so great that the effect of the discharge was eclipsed.

TABLE 9.2 SUMMARY OF UPSTREAM-DOWNSTREAM DIFFERENCES IN METALS CONCENTRATIONS

		No. of sites with			Difference* mg/kg			
		D<U	D=U	D>U	min	max	mean	median
Zinc	S	23	4	18	-1810	910	-33	-3
	A	14	1	17	-420	500	20	10
Cadmium	S	7	31	7	-2.8	10.8	0.2	0
	A	20	0	12	-6.1	6.4	0	0
Lead	S	19	3	23	-600	262	-12	10
	A	16	3	13	-230	190	-15	-1.5
Nickel	S	24	1	20	-228	65	-4.7	-6
	A	13	0	19	-21	102	3	1
Boron	S	25	2	18	-94	88	-2.7	-1.3
	A	13	1	18	-99	63	-1.3	2.5
Chromium	S	17	6	22	-57	17	-1.2	0
	A	14	2	16	-18	30	0.4	0.5
Vanadium	S	22	6	17	-12	27	0.2	0
	A	17	1	14	-22	24	0.6	-1
Copper	S	21	2	22	-1260	189	-27	0
	A	13	4	15	-840	22	-31	0
Silver	S	4	34	7	-9.9	32	0.7	0
	A	5	15	12	-2.2	5.2	0.5	0
Arsenic	S	22	3	20	-1.6	8	0.2	0
	A	13	3	16	-2.5	2.7	0.2	0.1

* negative value indicates downstream concentration lower than upstream concentration.

U = concentration at upstream station; D = concentration at downstream station; S = sediments; A = algae.

A second broad distinction can be made between the two sample types. Most metals are associated with fine sediment which may be moved at times of high flow, whereas *Cladophora* are sedentary algae. This distinction is clouded by the fact that contaminants do build up in sediments which are not eroded, and the fact that one of the sources of dissolved metals is sediment. Relative to sediments, *Cladophora* may also be short-term indicators of pollution; since metals can only be taken up by plants during their life span, whereas metals can build up in sediments over longer periods.

No evidence was found of a difference in metal concentrations between sampling stations. In many cases the measured differences in concentration were small and could have been caused by sampling or analytical error. The figures in Table 9.2 show that for all upstream-downstream sample pairs, the metal concentrations increased and decreased in approximately equal proportions. This was confirmed by a series of t-tests on upstream-downstream pairs, using transformed data so that the assumptions of homoscedasticity and normality were met. No significant differences were found except for silver, which was significantly higher in algae downstream of outfalls than upstream.

The data were also checked for any relationship between changes in algae samples and changes in sediment samples. A non-parametric test was used because the metal concentration differences were not normally distributed. There was no evidence of a relationship between concentration differences in the two sample types, although the largest increases and decreases were frequently at the same sites.

To investigate relationships between different metals, the upstream-downstream differences for each metal were compared with the differences for other metals. A non-parametric test was used because the differences were not normally distributed, and the results for sediment and algae samples are given in Table 9.3. The correlations were positive except for those with boron in algae samples. The two significant correlations (with chromium and copper) were both negative.

TABLE 9.3 CORRELATION BETWEEN UPSTREAM-DOWNSTREAM DIFFERENCES FOR DIFFERENT METALS

	Zn	Cd	Pb	Ni	B	Cr	V	Cu	Ag	As
Zn	**									
Cd	*A	**								
Pb	Sa	*a	**							
Ni	sA	*A	Sa	**						
B	Sa	*a	Sa	sa	**					
Cr	Sa	*a	SA	sA	SA	**				
V	SA	*A	SA	sA	sA	Sa	**			
Cu	Sa	*a	SA	sA	SA	SA	SA	**		
Ag	*A	*a	*a	*A	*a	*a	*a	*A	**	
As	Sa	*A	sA	sA	Sa	Sa	SA	Sa	*a	**
	Zn	Cd	Pb	Ni	B	Cr	V	Cu	Ag	As

S = significant correlation in sediment samples
s = not significant at 5% level
A = significant correlation in algae samples
a = not significant at 5% level
* = not tested

For sediment samples, increases in one metal were generally accompanied by increases in others, with the exception of nickel.

For algae samples, nickel increases correlated well with other metals, indicating that the nickel was entering the river in the dissolved state and not being adsorbed to sediments. There were not as many significant correlations between metal increases in algae samples as in sediment samples.

9.3 RELATIONSHIP BETWEEN BIOLOGICAL DIFFERENCES AND METAL CONCENTRATION DIFFERENCES

Metal concentration differences in sediment and algae samples were compared with BMWP score differences and comparative indices. The degree of association between changes measured using these methods was generally very low.

The only significant association found between biological parameters and metals in sediments was for nickel. Increased nickel concentrations downstream of the outfall were accompanied by increased BMWP scores.

For algae samples, scores downstream of the outfall increased with increasing concentrations of cadmium, boron and vanadium. No explanation was found for this unexpected result. Increases in zinc and nickel concentrations in algae were associated with decreases in comparative indices.

The lack of association between upstream-downstream differences in metals and corresponding biological differences was confirmed by examining data for sites where there were large increases in metals concentrations. These sites, at which some biological effect might be expected, were listed with their upstream and downstream BMWP scores. There was no evidence that a large increase in metals in sediments or algae was accompanied by a fall in BMWP score, even at sites where the upstream BMWP score was high. It was assumed that the observed metals levels were not sufficiently high to affect BMWP scores.

9.4 SEASONAL DIFFERENCES IN BMWP SCORES

Six sites were sampled at three month intervals from September 1986 to April 1988, so that seasonal differences could be assessed. Bad weather in January 1987 prevented sampling, and at other times sampling at some sites was not possible because the water was too deep. Although the resulting data set was incomplete and therefore difficult to interpret, some general observations can be made.

At four of the sites the range of BMWP scores at the upstream station was small in comparison with the range at the downstream stations. There was no seasonal impact pattern for these sites - the greatest impacts occurred at different times of year. At two of these sites there was generally very little impact, at the third a small drop in BMWP score was always detected, and at the fourth the impacts were variable.

At the two remaining sites there was a wide range of upstream BMWP scores, in both cases the result of one very low score for a usually high-scoring site, followed by some recovery. The upstream score at one site fell from 72 in July 1987 to 21 in January 1988, and at the other site from 47 in July 1987 to 13 in October 1987. Both these sites were of lower quality downstream of the outfall before the upstream score fell and in both cases this impact was subsequently eclipsed. It appears that at these sites an incident upstream of the outfall depressed the fauna to such an extent that the impact of the outfall could no longer be detected.

9.5 THE EFFECT OF CATCHMENT CHARACTERISTICS ON OUTFALL IMPACT

The upstream-downstream differences described above were highly variable. For both biological and metals observations, there were sites where the outfall was apparently damaging to water quality and sites where quality was higher downstream. Sites at which an impact was detected using metals data did not correspond to those at which a biological impact was detected. Overall, biological quality was significantly lower at downstream sampling stations than at upstream stations, but there was no such overall effect for any metals.

The effect of various catchment characteristics on the observed impacts was assessed by dividing the data set into categories based on measured characteristics. The characteristics considered were (sewered) catchment area, land use, observed contamination, and initial water quality. The upstream water quality categorisation also allows some assessment of the sensitivity of the methods used.

The biological measures used to assess the effect of catchment characteristics were BMWP scores and both comparative indices. Weighted BMWP scores were used in some cases, but taxa counts were not used as it had already been demonstrated that they detected the same impacts as BMWP scores.

All metals in sediments and algae samples were used in this part of the analysis, except silver, as it was infrequently detected, and cadmium in sediment samples for the same reason. Correlation between different metals had been demonstrated, but was not considered strong enough for any elements to be omitted.

9.5.1 The effect of catchment area on outfall impact

The monitored catchments were divided into five categories according to their area, as shown below:

AREA (ha)	< 1	1-10	11-50	51-100	> 100
No.of sites	8	19	12	4	4
mean area (ha)	0.69	5.38	18.5	64	220

Biological parameters and metals concentrations in each group were then analysed.

<u>Biological impact</u>

Paired comparison t-tests were carried out on the upstream and downstream BMWP scores in each group. Downstream scores were not significantly lower than upstream scores at sites less than 1 ha or sites between 1 and 10 ha. The 11 to 50 ha group downstream scores were significantly lower at the 10% level, and the 51 to 100 ha group score differences were significant at the 5% level. Downstream scores at sites over 100 ha were also significantly lower than upstream scores at the 5% level.

These results indicate that the biological impact increased with catchment area. A single classification anova was carried out on the score differences in each group. The mean BMWP score differences were as follows:

< 1	ha	-2.25
1-10	ha	0.05
11-50	ha	-3.83
51-100	ha	-4
> 100	ha	-9

The anova result did not confirm that catchment area affected outfall impact on BMWP scores. The conflicting results obtained by the two methods used indicate that the relationship between biological impact and catchment area is on the margins of statistical significance. There appears to be increasing impact with catchment area, but there is also wide variation within groups of sites of similar area.

The t-tests and anova were repeated for weighted scores, with similar results. There was no significant upstream-downstream reduction of scores at smaller sites, significantly lower downstream scores at large sites, but no evidence of an area effect when an anova was carried out on the score differences.

To determine whether there was any difference in the nature of the biological impact for different catchment sizes, the catchment area classification was marked on a plot of the two comparative indices (Fig. 9.1). Sites at which there was an apparent increase in quality downstream of the outfall tended to be the small sites. All except 1 of the smallest (<1 ha) sites had high species deficit indices. Sites with taxa missing downstream and no replacement groups were not the large sites as expected. Sites in the 1 to 50 ha groups showed a wide range of effects.

Impact on metals concentrations

Paired comparison t-tests were carried out on upstream-downstream metal pairs for sediment and algae samples. Log transformed concentrations were generally required to meet the assumptions of normality and homoscedasticity. Two-tailed tests were carried out on metals samples as there was no evidence of any change in concentration using the whole data set.

80 values of t were calculated, for 8 metals, 5 catchment area groups, and 2 sample types. Only two t values were greater than the tabulated values at the 5% level: zinc in algae samples was lower downstream than upstream at sites smaller than 1 ha and chromium in sediment samples was higher at downstream stations of sites between 51 and 100 ha. Two further t values indicated significant differences at the 10% level: for chromium in sediments at sites smaller than 1 ha, where downstream

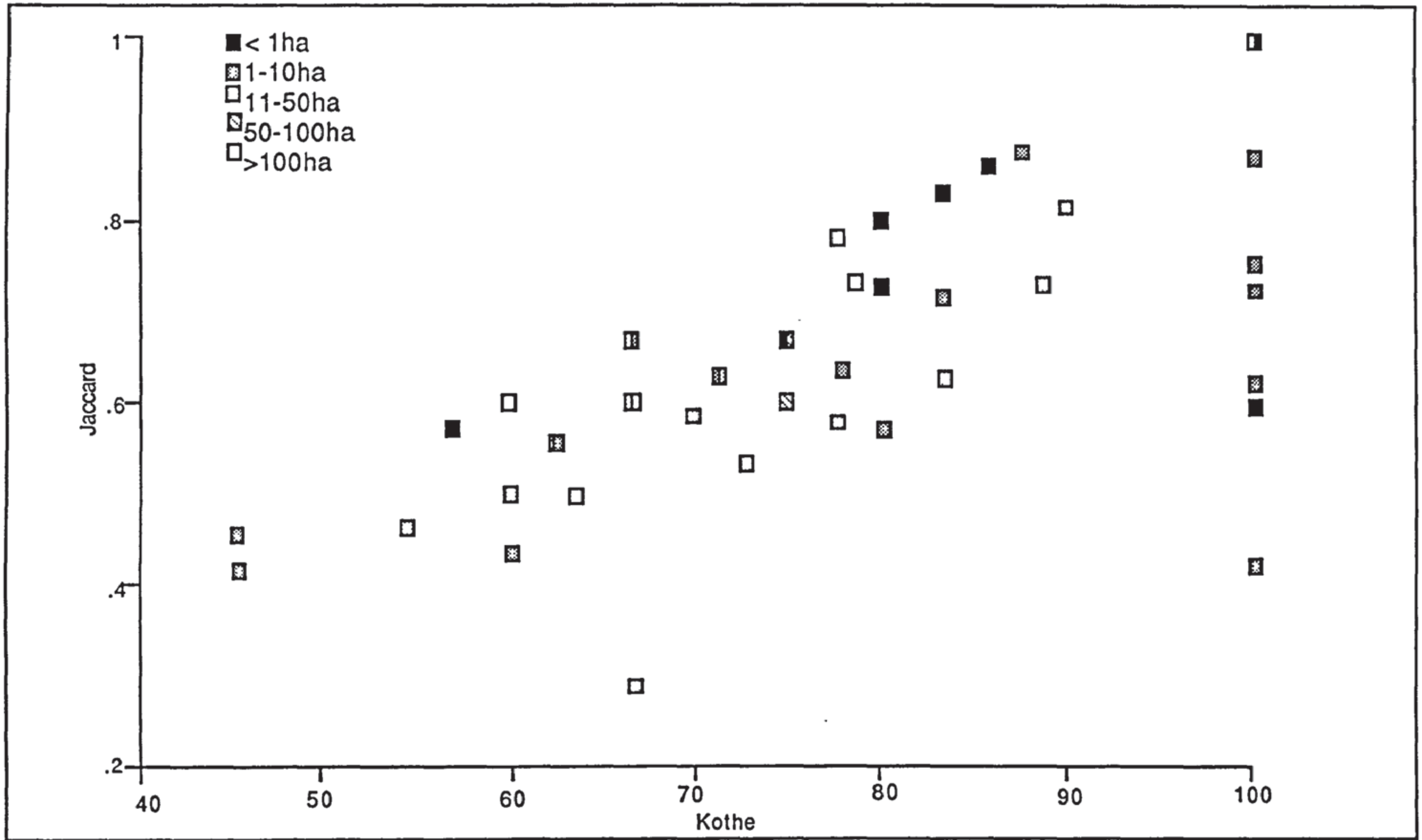


Fig 9.1 Biological impact and catchment area

concentrations were lower, and vanadium in sediments which was present in higher concentrations at downstream stations of 51 to 100 ha sites.

It was concluded that catchment area had no effect on changes in metals concentrations in sediments or algae downstream of outfalls.

9.5.2 The effect of catchment land use on outfall impact

The catchments were divided into four categories according to their land use, as shown below:

LAND USE	Residential	Industrial	Highway	Mixed
No.of sites	21	4	4	18

Sites were only placed in the residential, industrial and highway categories if these were the sole land use in the catchment. Sites which were predominantly residential but with industrial or commercial estates were classed as mixed. In this way the effect of the non-residential area, which was usually of considerable size as the mixed land use catchments were generally large, could be assessed.

Biological parameters and metals concentrations in each land use category were analysed.

Biological impact

Paired comparison t-tests were carried out on the upstream and downstream BMWP scores in each group. For residential and mixed land use sites the BMWP scores were not significantly lower at downstream stations. At the industrial and highway sites downstream BMWP scores were significantly reduced, at the 10% and 5% confidence levels respectively. Similar results were obtained using weighted scores.

A single classification anova was carried out on the BMWP score differences in each land use category. The mean BMWP score differences were as follows:

residential sites	-0.95
industrial sites	-8.25
highway drainage	-4.50
mixed land use sites	-2.44

The anova did not confirm the result of the t-tests: the variance within the categories was greater than that between categories.

To determine whether there was any difference in the nature of the impact for different catchment land uses, the land use was marked on a plot of the two comparative indices (Fig. 9.2). No clear patterns were apparent, although as expected the industrial sites generally had low index values. The highway sites were all on or close to the line representing taxa loss with no replacement downstream groups. However, there were only 4 highway sites and at one of these the taxa lists were identical upstream and downstream.

The biological results generally indicate that industrial and highway drainage is damaging to water quality, and that residential and mixed land use catchment effects are lower but highly variable.

Impact on metals concentrations

Paired comparison t-tests were carried out on upstream-downstream metal pairs for sediment and algae samples. Transformed data were generally used so that the assumptions were met, and the significance tests were all two-sided.

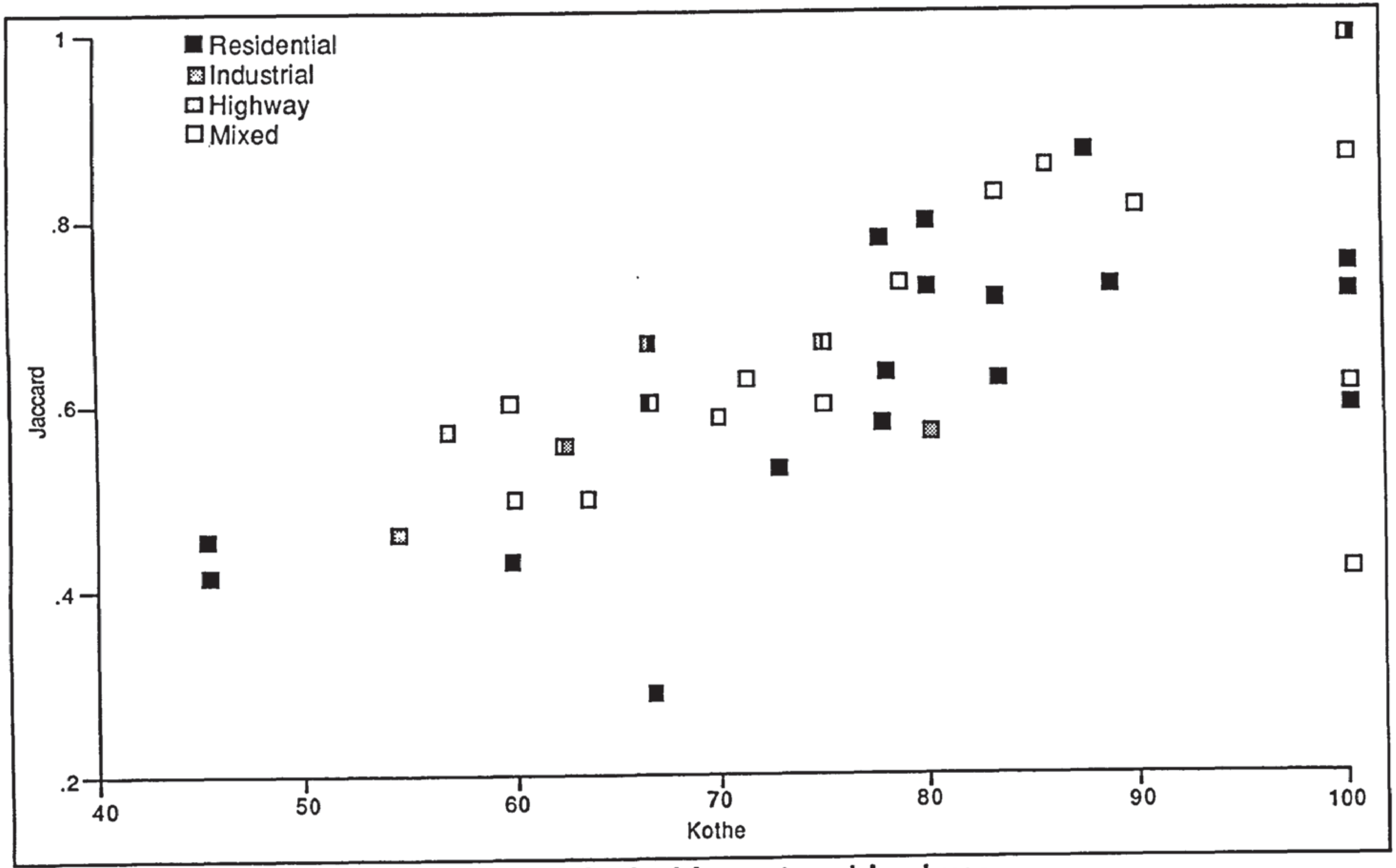


Fig 9.2 Biological impact and land use

The only significant differences found between upstream and downstream samples were for nickel, which was present in lower concentrations in sediments downstream of industrial site outfalls, and boron, which increased in concentration downstream of highway drain outfalls. With these two exceptions it was concluded that catchment land use did not affect metals concentrations in sediments and algae.

9.5.3 The effect of sewage and other contamination on outfall impact

Using the data summarised in Table 7.5, the sites were grouped according to observed discharges. Four groups were recognised: sewage contamination, industrial effluent, unidentified dry weather discharge, and 'clean'. Outfalls which had been operating in dry weather but appeared clean were classed as such. This resulted in the following grouping:

DISCHARGE	Sewage	Industrial	Unidentified	Clean
No.of sites	13	2	5	27

Because the outfalls were not continuously monitored, sites can not be classed as clean with great confidence. Significant sewage flows can generally be detected after the event, as was the case at one site, but intermittent illegal industrial discharges are more difficult to identify. It is also possible that observed dry weather discharges which were classed as clean contained clear, colourless and odourless pollutants.

Biological impact

Paired comparison t-tests were carried out on the upstream and downstream BMWP and WBMWP scores in each group. The values of t obtained are given in Table 9.4.

TABLE 9.4 VALUES OF t FOR UPSTREAM AND DOWNSTREAM SCORES AT SITES GROUPED BY OBSERVED DISCHARGES

DISCHARGE	SCORE	t	SIGNIFICANCE LEVEL %
sewage	BMWP	-1.3743	10
	WBMWP	-2.1361	5
industrial effluent	BMWP	-2.1429	-
	WBMWP	-4.4615	10
unidentified	BMWP	1.2665	-
	WBMWP	0.8733	-
clean/absent	BMWP	-1.9825	5
	WBMWP	-1.7224	5

Scores at sewage contaminated and 'clean' sites were significantly lower downstream of the outfall. WBMWP scores were significantly lower downstream of sites where industrial effluent was present. At sites with an unidentified dry weather discharge the scores were not lower downstream of the outfall.

Two single classification anovas were carried out on the score differences in the four contamination classes, for BMWP scores and WBMWP scores. Both tests gave significant results which confirmed that the discharge type affected biological impact. The mean score differences at the different sites were as follows:

	mean score difference	
	BMWP	WBMWP
sewage	-4.5	-9.7
industrial	-7.5	-29.0
unidentified	9.0	9.7
clean	-3.2	-3.9

The biological impacts were further investigated using a plot of the two calculated comparative indices (Fig. 9.3). Sites with low values of Kothe's Species Deficit (less than 70) contained a higher proportion of industrial and sewage contaminated discharges, with 'clean' systems accounting for 47% of the total. Sites with higher index values were predominantly (63%) 'clean'. Not all sites with sewage contamination suffered a deterioration in water quality downstream of the outfall, however, and at one sewage-contaminated site there were new taxa present downstream without loss of upstream groups.

Impact on metals concentrations

Paired comparison t-tests were carried out on upstream-downstream metal pairs for sediment and algae samples. Transformed data were used where necessary and the significance tests were two-sided.

No significant upstream-downstream differences in metals concentrations were found for sediments or algae in any of the discharge groups. It was concluded that metals concentrations were not affected by the observed discharges.

9.5.4 The effect of initial biological quality on detected outfall impact

To investigate the effect of upstream water quality on the biological impact detected, the measured impacts were compared

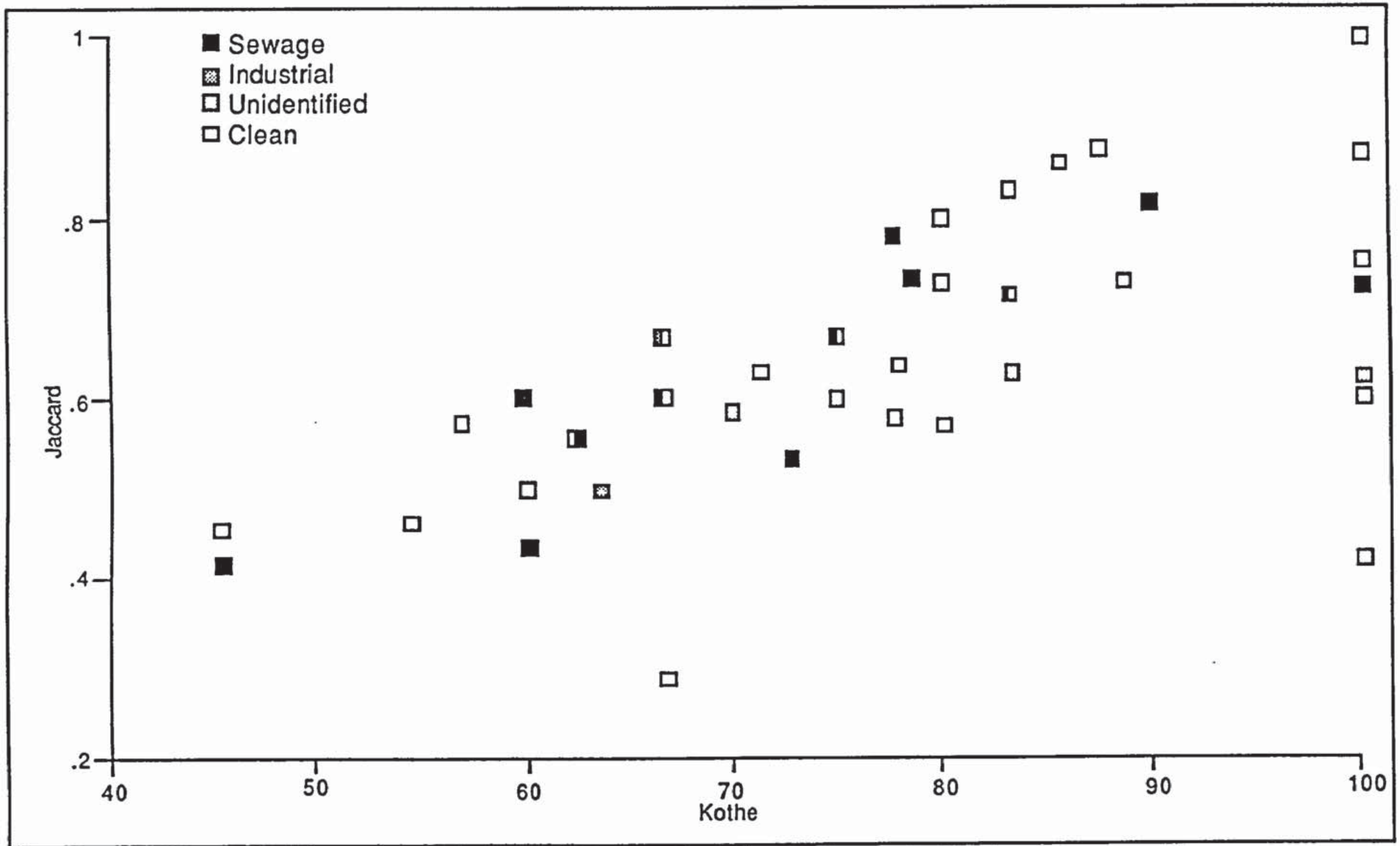


Fig 9.3 Biological impact and contaminated discharges

with the watercourse NWC classifications and upstream BMWP scores.

Sites were first grouped according to their NWC class:

Class 1B	20 sites
Class 2	17 sites
Unclassed	10 sites

Paired comparison t-tests were carried out on the upstream and downstream scores for each group of sites. For class 1B sites there was no significant reduction of BMWP scores at downstream stations, but the reduction in WBMWP scores was significant at the 10% level. No significant score reductions were found for class 2 sites. At unclassed sites, however, the reductions in BMWP and WBMWP scores were significant at the 10% and 5% levels respectively.

The unclassed watercourses are minor tributaries which are too small and too numerous for inclusion in the NWC classification scheme. The biological results indicate that it is these small streams which suffer the greatest impact from surface water discharges. In addition and as expected, class 1B watercourses are more likely to be adversely affected by discharges than class 2 watercourses.

The association between upstream quality and detected impact was further investigated by comparing upstream BMWP scores with BMWP score differences (Fig. 9.4). The largest score reductions are at sites where the initial water quality is highest, and downstream increases in scores are more common at sites where the initial quality is poor. The correlation between upstream BMWP scores and upstream-downstream score differences was significant at the 1% confidence level.

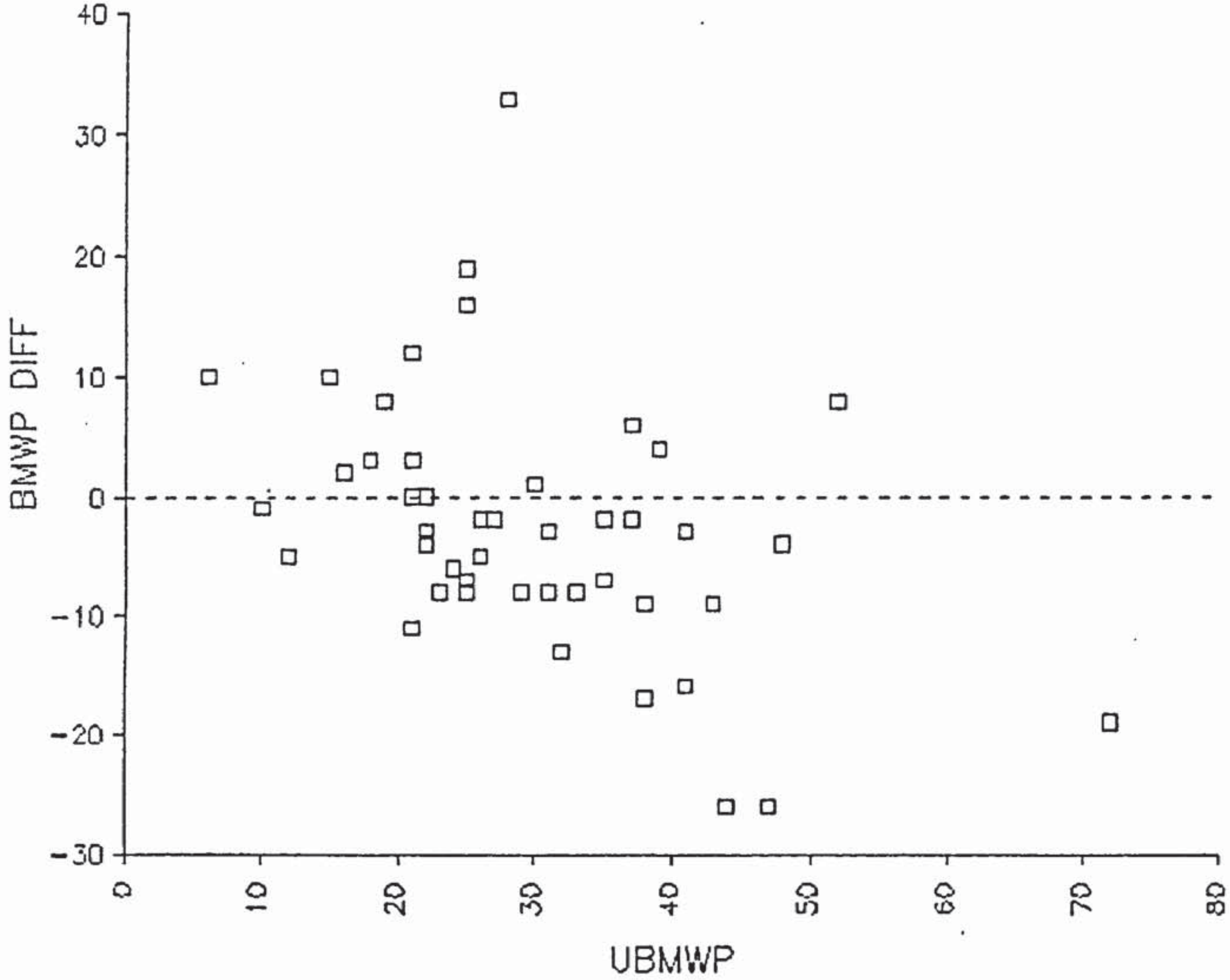


Fig. 9.4 Biological Impact and Upstream Water Quality

80% of moderate and good biological quality sites (upstream BMWP score over 25) had lower BMWP scores downstream of the outfall than upstream. For unsatisfactory and poor quality sites (BMWP score below 25) 45% of downstream scores were lower and 45% were higher than upstream of the outfall.

Initial water quality is clearly an important factor in the consideration of outfall impact. Poor quality streams were not adversely affected by surface water discharges, but at most of the better quality watercourses studied there was deterioration in quality downstream of the outfall.

9.5.5 The effect of other catchment characteristics on outfall impact

Degree of urbanisation upstream of the sites studied and their drainage area ratios were also used to group sites for impact assessment. No evidence was found of an effect related to degree of urbanisation for either biological scores or metals in sediments and algae.

Drainage area ratios (DAR) were used to classify sites into two groups: sites with DAR <1 and those with DAR $>=1$. BMWP and WBMWP scores were not significantly reduced at downstream stations of the first group, which contained 34 sites. 13 sites belonged to the second group, in which BMWP and WBMWP scores were significantly (1%) lower downstream of the outfall than at upstream stations. This supports earlier observations that biological impact increased with catchment area (section 9.5.1) and was greater for small streams (section 9.5.4).

9.6 SUMMARY

Biological scores and metals concentrations in sediments and algae indicated a variety of water quality effects downstream of outfalls. Upstream-downstream differences were generally inconsistent and often small.

9.6.1 Summary of overall upstream-downstream differences

Most taxa counts, BMWP scores and weighted scores were lower at sampling stations downstream of outfalls than at upstream stations. Kothe's Species Deficit and Jaccard's coefficient indicated different types of biological impact, which when compared with BMWP scores showed that most score reductions were the result of upstream taxa being replaced by lower scoring groups downstream.

No evidence was found of a difference in metals concentrations in sediments or algae upstream and downstream of the studied outfalls, and concentration changes in sediments were not reflected by changes in algae.

There was little association between upstream-downstream metals differences and corresponding BMWP scores and comparative indices.

9.6.2 Summary of the effects of catchment characteristics on outfall impact

Catchment area influenced the biological impact of outfalls, with the greatest impact for large catchments and no significant impact for catchments under 10 ha. Classification of sites into residential, industrial, highway, and mixed land use groups showed that industrial and highway catchments were most damaging to water quality. Neither area nor land use significantly affected metals concentrations.

Systems with observed sewage contamination had a significant biological impact, but not all such sites were damaging to water quality and systems which were apparently clean also significantly affected water quality. Metal concentrations were not affected by discharges.

Discharges to small watercourses not included in the NWC classification scheme were more damaging to water quality than discharges to class 1B rivers, which in turn were more likely to suffer than class 2 rivers. The high impact on unclassfied watercourses was attributed to their small size, and this was confirmed by comparison of impacts for sites with different drainage area ratios.

Initial water quality was an important factor in outfall impact. At sites where water quality was poor (BMWP score below 25), quality was as likely to improve as deteriorate downstream. At moderate and good biological quality sites, 80% suffered a reduction in BMWP score downstream of the outfall.

The integrated effects of catchment characteristics and their role in determining discharge impact are considered in more detail in Chapter 11, where the observations summarised above are combined to derive a simple planning tool.

CHAPTER 10

MACROINVERTEBRATE COMMUNITY CHANGES DOWNSTREAM OF OUTFALLS

The purpose of this chapter is to describe changes in macroinvertebrate communities upstream and downstream of outfalls where an impact was detected, and to identify any taxa commonly affected by discharges.

The effect of a series of outfalls on benthic macroinvertebrates is also investigated by looking at the changing community structure for two groups of outfalls.

10.1 SITES FOR FURTHER ANALYSIS

10.1.1 Selection of sites

The sites selected for more detailed analysis of upstream-downstream differences were restricted to those of moderate (BMWP score over 25) or higher upstream quality. This included most of the high impact sites and ensured a reasonable upstream macroinvertebrate community.

26 sites had upstream BMWP scores over 25, and at 21 of these the BMWP score was lower downstream. The sites with score differences of 10 or more were selected for more detailed analysis. 7 sites were thus selected: 6 with downstream scores lower than upstream and 1 with a higher score downstream.

10.1.2 Characteristics of selected sites

The basic characteristics of the 7 selected sites, referred to as 'worst impact' sites, are given in Table 10.1.

Site 1 was a small recently built residential estate of privately owned detached and semi-detached houses. The site was visited on several occasions, and there was usually a dry weather sewage flow at the outfall. The substratum downstream of the outfall had a higher proportion of silt than upstream.

Site 2 was a series of small road drain outfalls close to a motorway junction, and included an outfall from a hypermarket construction site. The latter outfall was discharging a white, turbid, frothy and odourless effluent which caused considerable silting for 5m downstream.

Site 5 was a small residential catchment with an apparently clean dry weather flow, and no silting observed downstream of the outfall.

Site 9 was also residential, but with a dry weather sewage flow and both silting and scour immediately downstream of the outfall.

Site 15 was an outfall for an airport runway drain. There was a strong odour of solvents at the outfall and excessive weed growth downstream, particularly of water cress. The weed growth made it difficult to assess the extent of silting but none was apparent.

Site 30 was a twin outfall serving a new residential estate and an adjacent industrial area. There was extensive silting downstream and a dry weather sewage flow in one pipe at the time of sampling.

Site 46 was a group of 3 small outfalls draining a new executive housing estate which was still partly under construction. There was an apparently clean dry weather flow at two of the outfalls, and silting caused by building materials at the third. There was some evidence of silting downstream, as the fine sediment on the stream bed was richer in clay, and there was building rubble dumped in the stream at the downstream stations.

TABLE 10.1 BASIC CHARACTERISTICS OF 'WORST IMPACT' SITES

Site No.	Area ha	Land use	DAR %	Contamination	BMWP diff.
1	5.3	Residential	0.26	sewage	-26
2	5.3	Highway/ commercial	0.17	white turbid DWF	+33
5	4.4	Residential	0.19	'clean' DWF	-26
9	11.4	Residential	0.26	sewage	-17
15	19	Airport	0.71	solvents	-16
30	115	Residential/ industrial	2.46	sewage	-19
46	6.9	Residential	0.76	'clean' DWF	-13

10.2 MACROINVERTEBRATE COMMUNITY CHANGES

10.2.1 Changes at 'worst impact' sites

The taxa lost and gained and the total number of organisms per sample are listed in Table 10.2. Oligochaete worms are not included as they were always present upstream and downstream of the outfall and usually in very large numbers.

With the exception of site 9, where there was a small rise in the number of organisms, and site 2 at which quality apparently

TABLE 10.2 CHANGES IN MACROINVERTEBRATE COMMUNITIES DOWNSTREAM OF 'WORST IMPACT' OUTFALLS

Site No.	Taxa lost downstream	Taxa gained downstream	Organisms U	Organisms D
1	Hydrobiidae Ancylidae Ephemerellidae Hydropsychidae Tipulidae	Sialidae	239	143
2	None	Hydrobiidae Glossiphonidae Erpobdellidae Asellidae Baetidae Elminthidae Chironomidae	65	235
5	Sphaeriidae Glossiphonidae Erpobdellidae Asellidae Baetidae Ephemerellidae	None	41	13
9	Ephemerellidae Elminthidae Dytiscidae	Erpobdellidae	121	152
15	Planariidae Planorbidae Ancylidae Erpobdellidae Dytiscidae	Baetidae Chironomidae	199	49
30	Heptageniidae Dytiscidae Limnephilidae	Erpobdellidae	49	39
46	Lymnaeidae Dytiscidae Tipulidae	None	243	174

Organism counts per 0.2 m^2 bed, excluding Oligochaeta improved, the organism counts as well as the BMWP scores were lower at the downstream stations.

In most cases, the fall in BMWP score was due to the loss of relatively high-scoring taxa, such as Trichoptera (caddis) and Ephemeroptera (mayflies), and as these were found in small numbers the basic community structure was not generally changed. This is illustrated for Site 1 in Fig. 10.1. Here at both the upstream and downstream stations the dominant taxa were chironomids and molluscs, but upstream there were also caddis and mayflies. There were relatively fewer *Gammarus* and *Asellus*, and overall numbers fell from 239 to 143 in $0.2m^2$ bed (excluding oligochaetes).

At site 15, the airport runway drainage site with strong solvent odours at the outfall, the community structure was changed significantly. The dominant group upstream of the outfall was molluscs, principally Lymnaeidae; but these were much reduced downstream with the result that crustacea, particularly *Gammarus* became dominant (Fig. 10.2). The total number of organisms present fell from 199 to 49 in $0.2m^2$ bed (excluding oligochaetes).

Without knowledge of discharge chemistry and frequency it is difficult to fully interpret the macroinvertebrate changes. The loss of high-BMWP-scoring taxa at most sites indicates that the discharges may be organic, as the system is based on organisms' tolerance to organic pollution. There were sites, however, at which sewage was present but sensitive taxa such as mayflies were apparently not affected. Suspended solids which interfere with invertebrates' feeding or attachment mechanisms are also likely to have an effect, and may be responsible for changes at apparently sewage-free sites.

At site 2, where there was a significant increase in community diversity and BMWP score, no satisfactory explanation for the

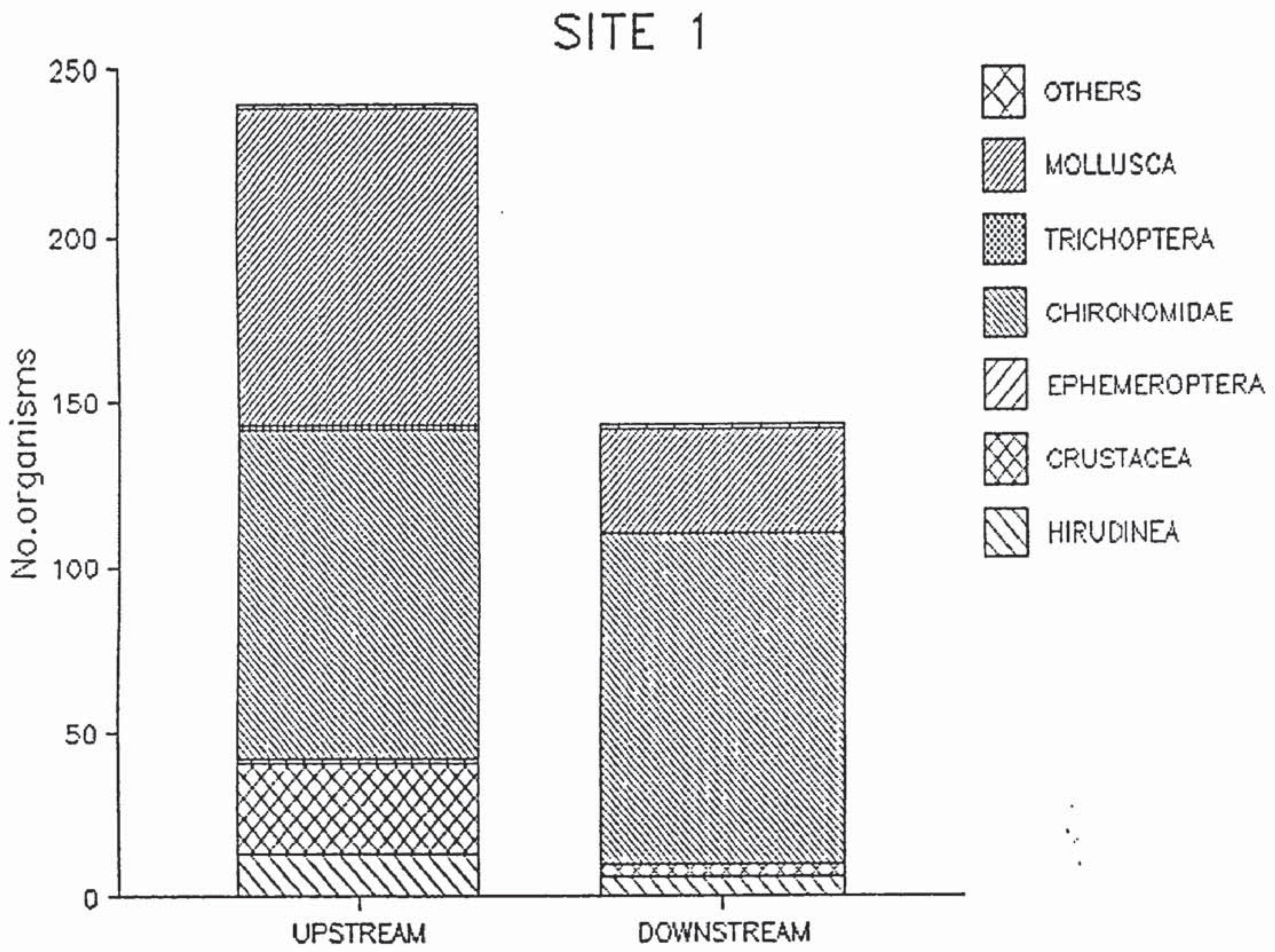


Fig. 10.1 Macroinvertebrate Community Structures at Site 1

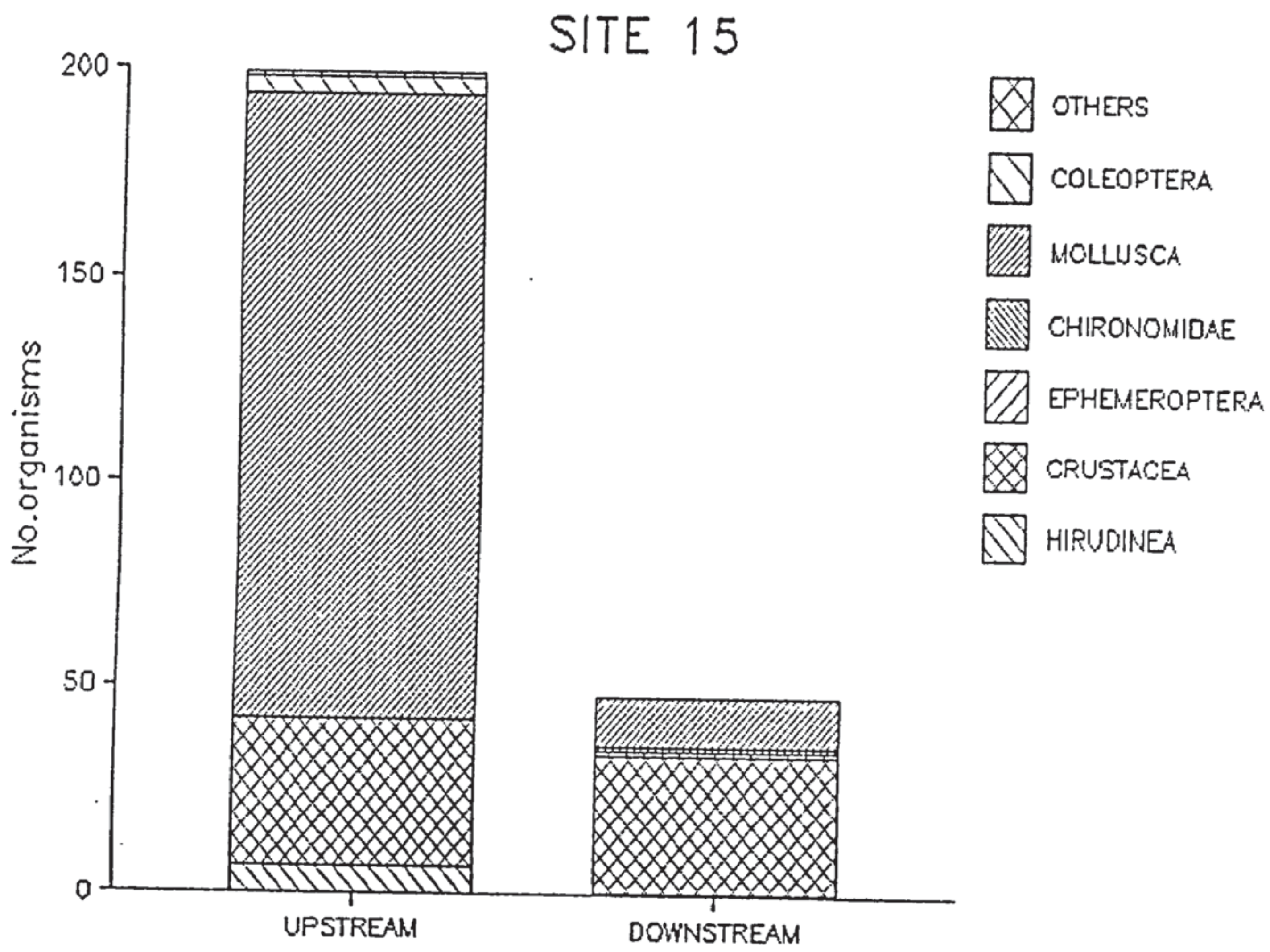


Fig. 10.2 Macroinvertebrate Community Structures at Site 15

changes was found. It is possible that the upstream fauna were supressed locally by an unknown factor, but such rapid recovery seems unlikely.

The taxa most commonly missing downstream of outfalls are identified in the following section.

10.2.2 Commonly disappearing taxa

Taxa which disappeared downstream of the outfall at the 7 'worst impact' sites are listed in Table 10.3 with the number of sites at which they were present upstream, and the number of sites at which they disappeared downstream of the outfall. The most commonly disappearing taxa were mayflies, beetles and caddis flies.

On the strength of the observations for the 'worst impact' sites, further sites were considered. The mayfly, beetle and caddis fly presences and disappearances at all of the 26 sites with BMWP scores over 25 were counted and are summarised in Table 10.4.

Consideration of more sites revealed that it was beetles and caddis which were most frequently missing downstream of outfalls. These taxa also infrequently appeared at downstream sampling stations where they were not present upstream. The exception to this general rule was cased caddis, which was found at more downstream than upstream stations.

Mayflies of the family Ephemerellidae were not as badly affected as had been indicated by the 'worst impact' sites. They were never found at downstream stations unless they were present upstream. There was no evidence that their numbers were affected by outfalls: at sites where they were present both upstream and downstream, it was in similar numbers.

TABLE 10.3 UPSTREAM PRESENCES AND DISAPPEARANCE OF TAXA AT 'WORST IMPACT' SITES

TAXON	Upstream presences	Downstream disappearances
Planariidae	1	1
Hydrobiidae	1	1
Lymnaeidae	5	1
Planorbidae	1	1
Ancylidae	4	2
Sphaeriidae	3	1
Glossiphonidae	6	1
Erpobdellidae	2	2
Asellidae	5	1
Baetidae	3	1
Heptageniidae	1	1
Ephemerellidae	4	3
Dytiscidae	4	4
Elminthidae	1	1
Hydropsychidae	2	2
Limnephilidae	1	1
Tipulidae	3	2

TABLE 10.4 UPSTREAM PRESENCE AND DISAPPEARANCE OF MAYFLIES, BEETLES AND CADDIS FLIES AT GOOD QUALITY SITES

TAXON	Upstream presences	Downstream disappearances	Disappearance rate %
Baetidae	13	6	46
Heptageniidae	1	1	100
Ephemerellidae	12	3	25
Haliplidae	2	2	100
Dytiscidae	8	8	100
Elminthidae	5	4	80
Hydropsychidae	9	7	78
Limnephilidae	1	1	100

Physical effects, particularly silting, may be responsible for the consistent loss of caseless caddis and beetles, neither of which are particularly sensitive to organic pollution. It is surprising that the effects of organic pollution were not more apparent, particularly at sites where sewage contamination was observed.

10.3 COMMUNITY CHANGES OVER A STREAM REACH

Two stretches of stream where there was more than one outfall were chosen for investigation of the effect of a series of outfalls on the macroinvertebrate community. Neither of these stretches contained any 'worst impact' outfalls.

10.3.1 Whetstone Brook

The Whetstone Brook close to the M1 motorway in Leicestershire is shown in Fig. 10.3. There were two outfalls (site numbers 28 and 29) on the stretch of brook shown. Approximately 2 km upstream of site 28 the brook received drainage from the motorway, but was otherwise free from major discharges.

Site 28 was a twin outfall draining 6.4 ha of the adjacent motorway and a 4.75 ha industrial estate. The pipe from the industrial estate was discharging in dry weather at the time of sampling, but although slightly oily the discharge appeared relatively clean.

Site 29 was an outfall from a 4.69 ha housing estate, still partly under construction. This outfall was also operating in dry weather, and the discharge was oily and smelled strongly of sewage.

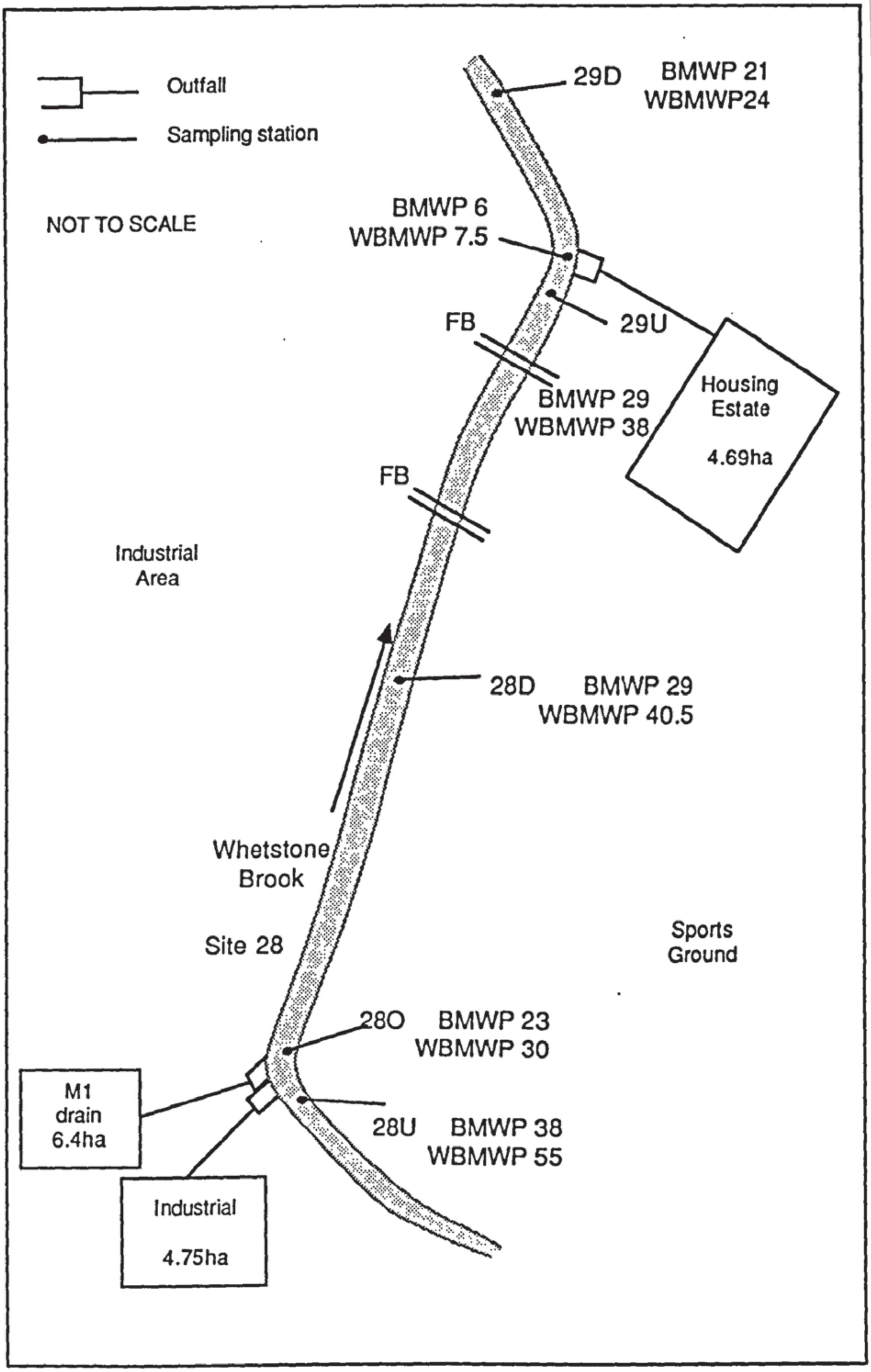


Fig 10.3 Whetstone Brook

At both sites there was silting for about 10 m downstream of the outfall.

The BMWP scores and WBMWP scores are given on Fig. 10.3, and BMWP scores are also shown in Fig. 10.4. Upstream of the first outfall at station 28 U the scores were the highest of the six sampling stations, and the biological quality was moderate. At the outfall station 28 O, the scores fell and then recovered slightly further downstream at station 28 D. Between station 28 D and station 29 U there was little change. At the outfall station 29 O, however, the scores fell to their lowest values and the quality at this point was unsatisfactory. By station 29 D there was some recovery and the quality improved from unsatisfactory to poor.

For both these outfalls there was a clear pattern of immediate high impact at the outfall, followed by recovery downstream. In this case the impact of the second outfall affected the stream before recovery from the first was complete, thus further damaging water quality. This was also reflected in the numbers of organisms present and the changing macroinvertebrate community structure (Fig. 10.5, oligochaeta excluded).

The basic community structure at station 28 O was the same as at station 28 U, but organism numbers were much reduced and so was diversity within the groups, as illustrated in Fig. 10.5. At stations 28 D and 29 U molluscs were present, whereas they were absent from other stations. There was also an increase in the amount of *Cladophora* at these stations, probably caused by increased nutrient concentrations downstream of the first outfall.

The results from this watercourse illustrate the effects of a series of outfalls on water quality when there is not sufficient distance between them for the stream to recover. Outfalls to urban streams are commonly close together, and this was one of the reasons why, in this study, it was not usually possible to

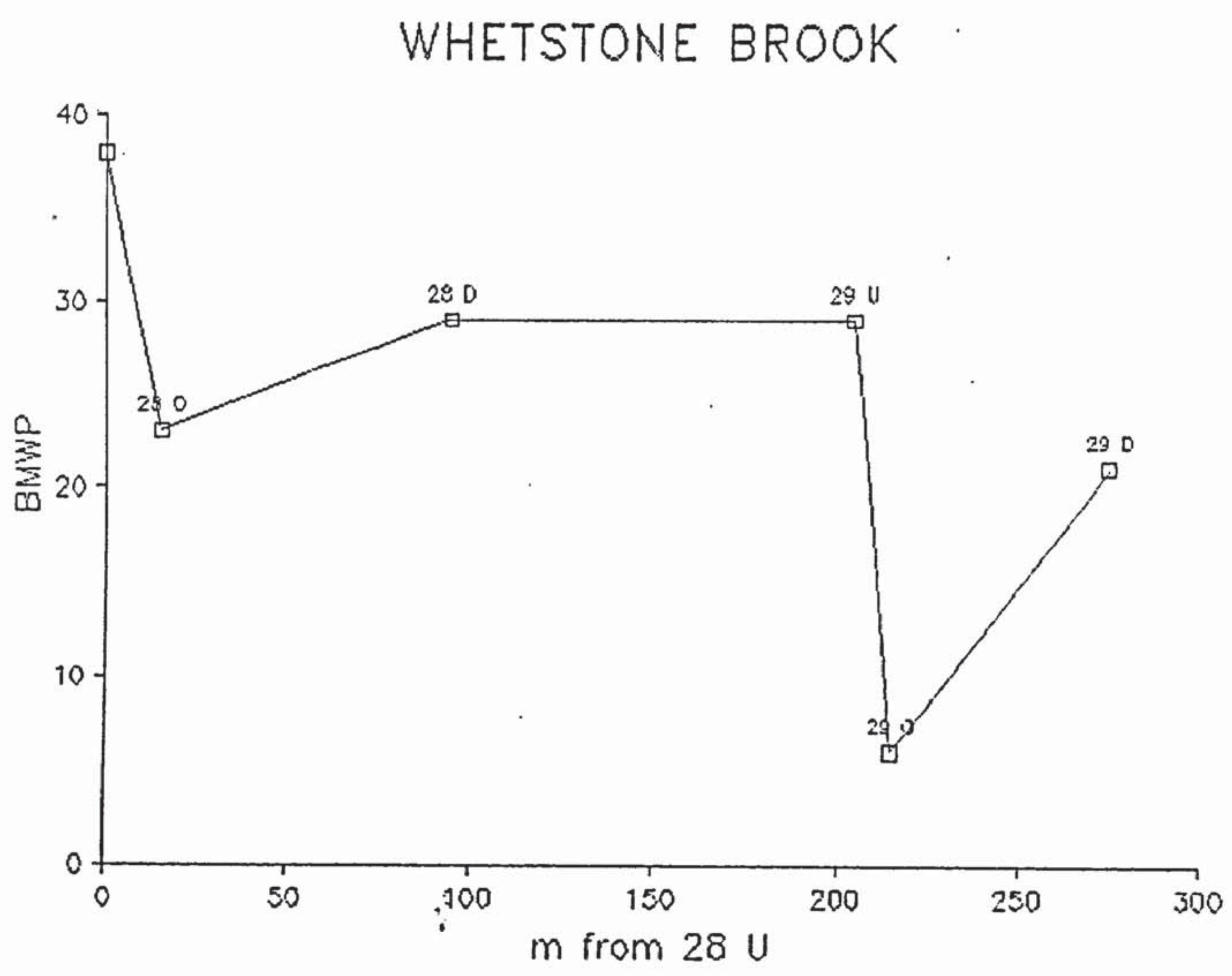


Fig. 10.4 BMWP Scores for Whetstone Brook

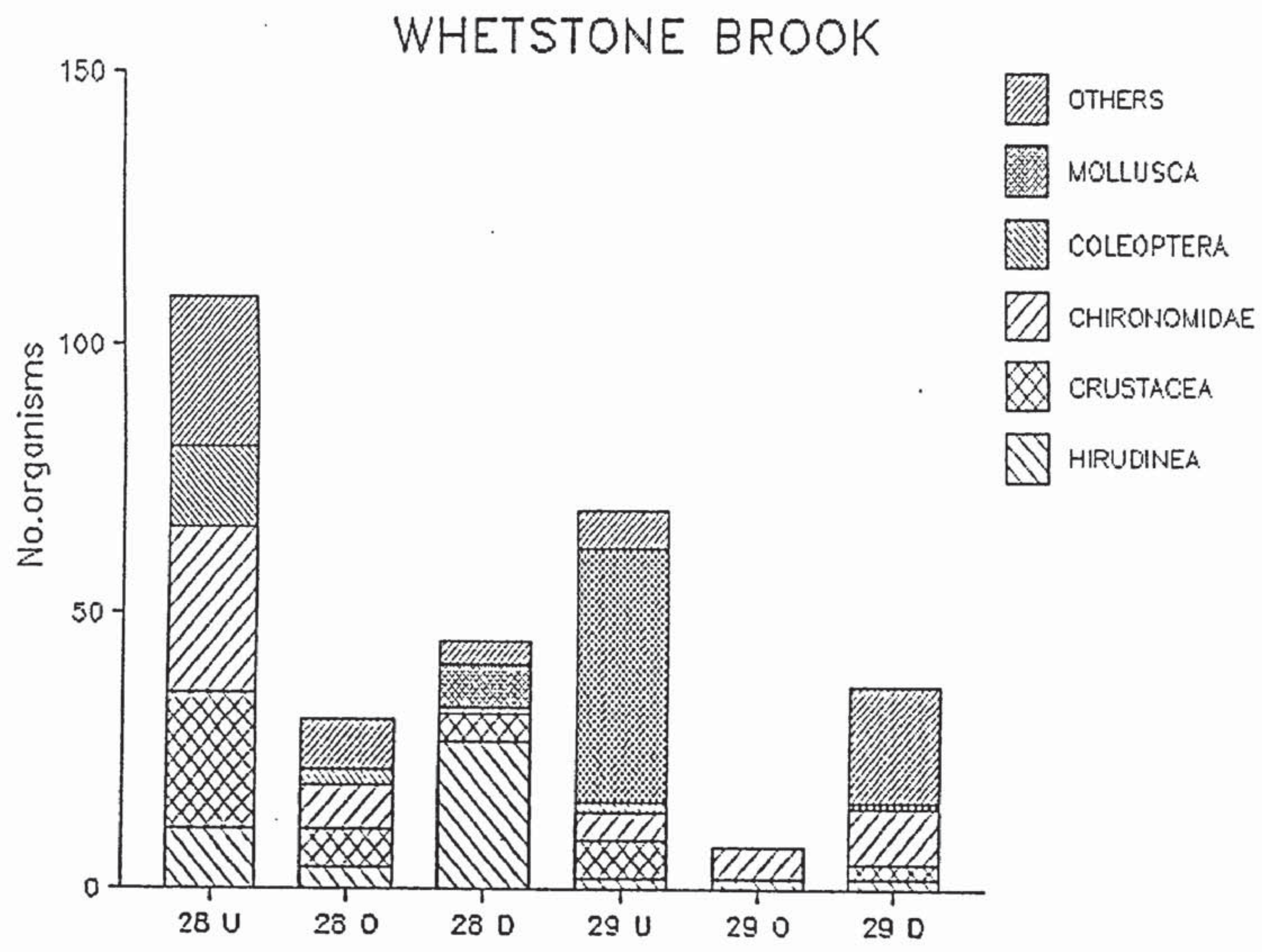


Fig. 10.5 Macroinvertebrate Community Structures in Whetstone Brook

sample more than 100m downstream of individual outfalls.

It is not possible, however, to determine whether the effect in this case was cumulative, as each of the individual outfalls appeared to have a measurable impact. The pattern of high initial impact followed by recovery at sites 28 and 29 was not common, and at most of the other outfalls studied there was no evidence of recovery at the downstream station (see section 9.1.1).

10.3.2 Griffins Brook

The second stream reach considered in detail was Griffins Brook, Birmingham, where it is crossed by the Bristol Road in Bournville, as shown in Fig. 10.6. Four outfall sites (numbers 20, 21, 22 and 23) were sampled on this reach.

Site 20 drained a 1.95 ha estate of student flats. No contamination was recorded at the outfall, and there was no silting downstream.

Site 21 was a group of outfalls from a 12 ha residential estate and 7.4 ha of the Bristol Road. These outfalls were close to and under the road bridge, which was too low for access, so the outfall station for these was the point where the brook emerged from the bridge on the downstream side of the Bristol Road. None of the visible outfalls were discharging in dry weather. There was some silting at the outfall mouths but its extent was difficult to assess because of the road bridge. There was no dumping close to the bridge. Sites 22 and 23 were outfalls draining blocks of recently built low-rise flats and adjoining private roads. There was no evidence of contamination or significant silting at site 22, but at one of the pair of pipes making up site 23 there was an apparently clean dry weather discharge and considerable silting for about 3 m downstream.

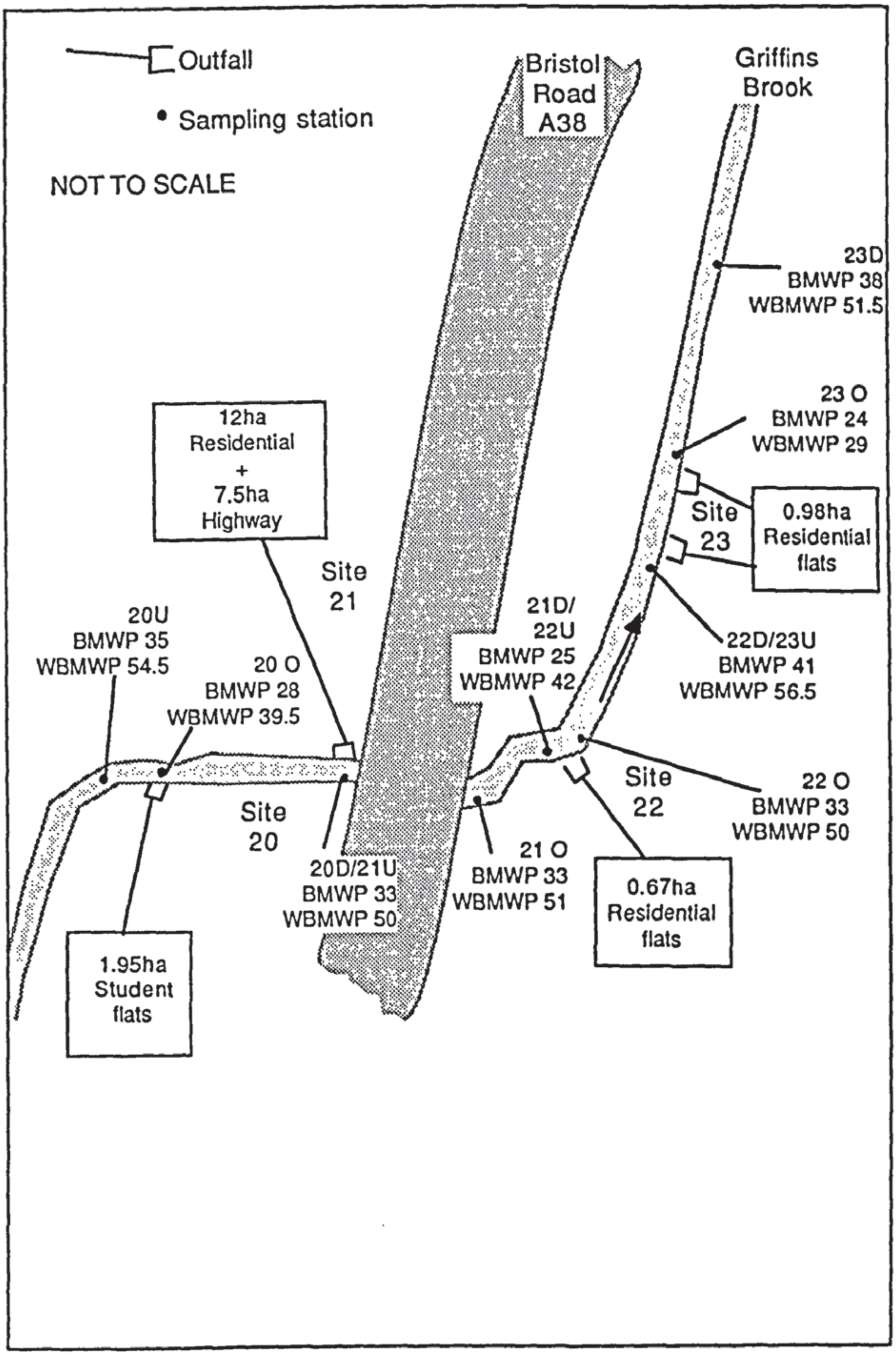


Fig 10.6 Griffins Brook

The BMWP and WBMWP scores are shown in Fig. 10.6 and the BMWP scores in Fig. 10.7. Unlike the previous stream reach described, there was no consistent pattern of change in the scores. Fluctuations between stations did not seem to be related to the presence of the outfalls, neither did the scores seem to be affected by the proximity of the major road. At sites 20 and 23 there was a fall in scores at the outfall followed by a recovery, but at site 22 there was an increase in score at the outfall. The score at the outfall station for site 21 was the same as the upstream score, but any immediate impact would have been missed at this site because the road bridge prevented sampling at the outfalls themselves.

The community structures and organism counts for Griffins Brook are illustrated in Fig. 10.8. The structures and counts were similar for all the sampling stations, with all the upstream groups represented at the downstream stations. Trichoptera, which were identified in section 10.2.2 as one of the most likely taxa to disappear downstream of outfalls, were present at all stations except 23 O.

These results indicate that for a series of outfalls which have only small individual impacts, there is no measurable cumulative effect on water quality. The water quality downstream of the last of the outfalls is similar to the quality upstream of the first outfall.

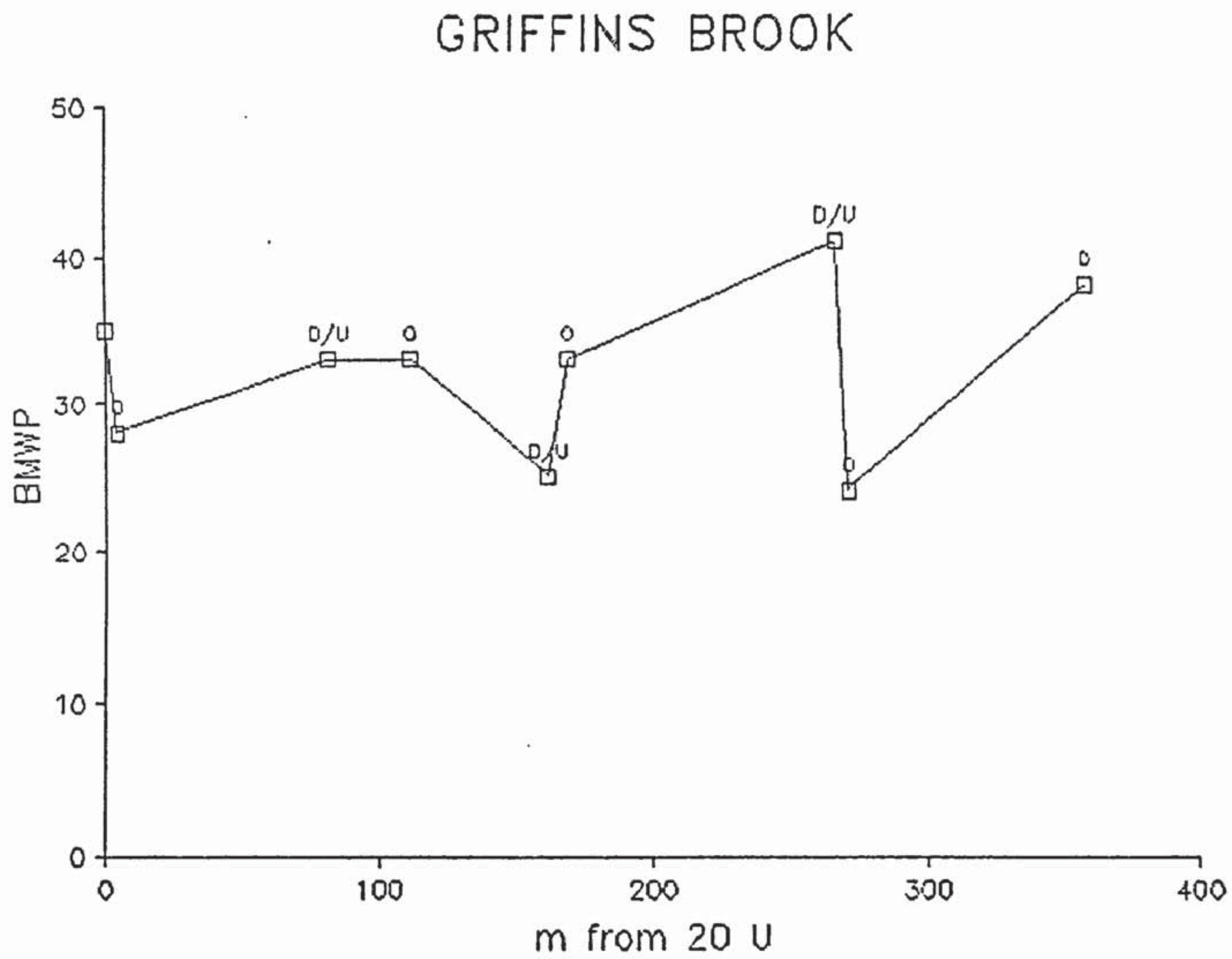


Fig. 10.7 BMWP Scores for Griffins Brook

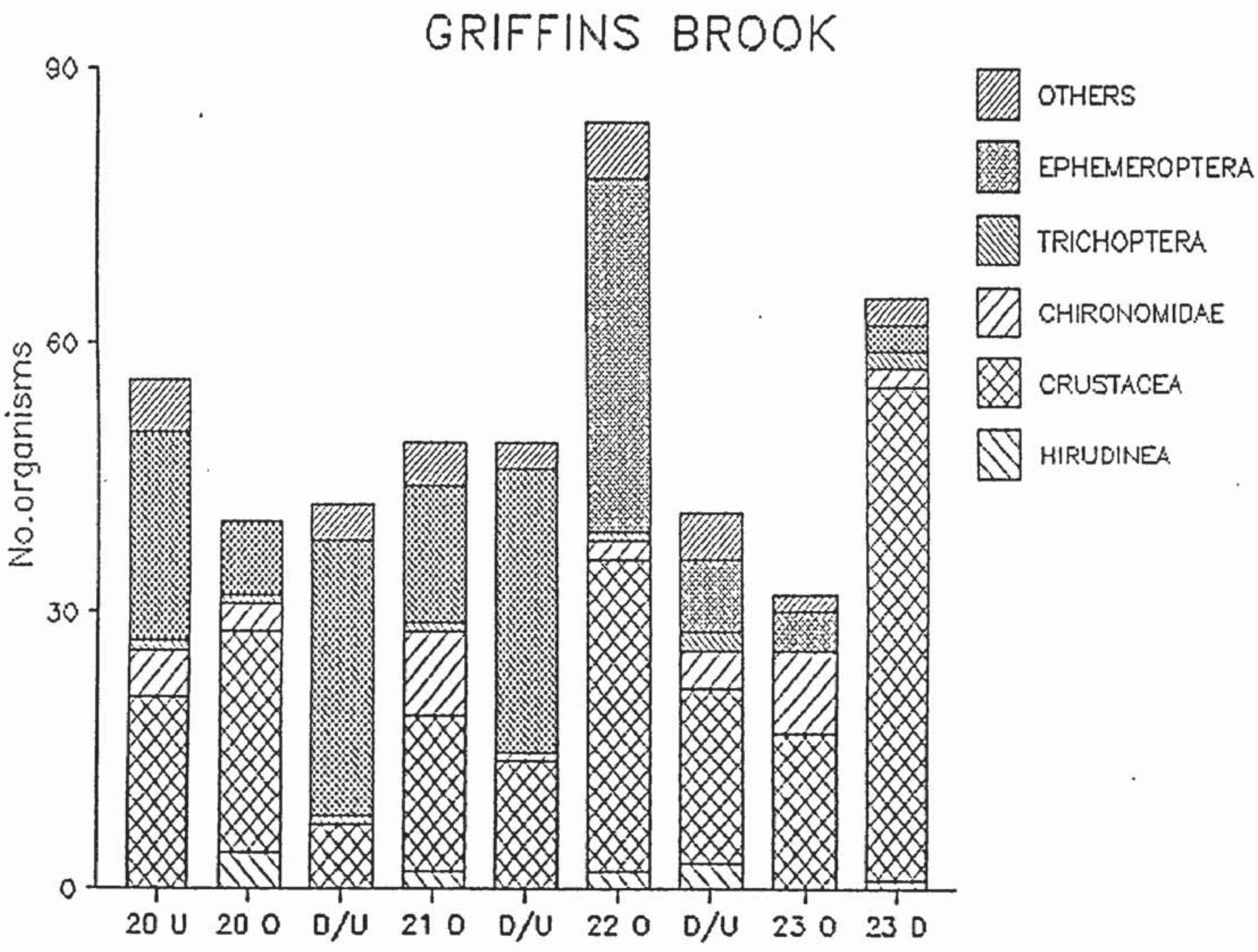


Fig. 10.8 Macroinvertebrate Community Structures in Griffins Brook

10.4 SUMMARY

Closer examination of results from sites where initial water quality was high showed that reduced BMWP scores downstream of outfalls were usually caused by the loss of high-scoring taxa such as caddis and mayflies. As these taxa were present in small numbers the overall macroinvertebrate community structure was not much changed. There were sites, however, where the community structure was changed significantly.

The loss of high-scoring taxa indicates that the cause may be organic pollution, but there were sites where sewage was present and sensitive taxa were not affected. The taxa which most commonly disappeared downstream of outfalls were beetles and caddis, possibly because of silting caused by suspended solids discharges.

The effects of a series of discharges on water quality were illustrated using two examples. The first showed how water quality could deteriorate when a second outfall to a stream had an effect before the stream had recovered from a discharge further upstream. The second did not indicate any cumulative effect as a result of discharges from four outfalls which had small individual impacts.

CHAPTER 11

DERIVATION OF GUIDELINES FOR IMPACT ESTIMATION

The purpose of this chapter is to describe how the survey results were used to produce tentative guidelines for estimating discharge impact from knowledge of simple, readily determined catchment characteristics. Further work to improve the reliability of the guidelines is recommended, and the application of the method is discussed.

In Chapter 9, biological impacts were identified and it was demonstrated that they were related to characteristics of the sewered catchment and to receiving stream quality. No consistent impacts were detected on levels of metals in algae or benthal sediments. In Chapter 10, it was shown that a series of small outfalls to a relatively clean brook had no detectable cumulative effect on water quality. The combined effects of receiving stream quality and catchment characteristics on biological impact are considered below.

11.1 HEIRARCHICAL CLASSIFICATION OF SITES

Further statistical analysis of the influence of catchment characteristics on discharge impact is complicated by the existence of relationships between the catchment characteristics and by gaps in the data set. Factors found to influence impact were, in likely order of importance:

Receiving stream quality
Drainage area ratio
Sewered catchment area
Catchment land use

These factors are all interrelated: for example, small catchments are unlikely to have mixed land uses; and stream quality may be influenced by DAR, which reflects the size of the total catchment area. Sewered catchment area and DAR are particularly difficult to separate, and their relative importance could only be assessed with any confidence from a larger data set. Rather than attempt to separate such factors, their effects were assessed using a heirarchical classification system for the study sites (Fig. 11.1). This approach had the added advantage that classes of site which required further study were easily identified.

```
                         BIOLOGICAL QUALITY
                         OF RECEIVING STREAM
                                  |
        --------------------------|-----------------------------
        |          |         |          |          |           |
        |          |         |          |          |           |
unsatisfactory   poor    moderate     good     very good   excellent
                             |
                             |
                             |
                         CATCHMENT
                           AREA
                             |
        ---------------------|----------------------
        |         |          |          |          |
        |         |          |          |          |
      <1ha      1-10       11-50      51-100     >100ha
                                        |
                                        |
                                        |
                                     LAND USE
                                        |
                 -----------------------|-------------------
                 |              |               |          |
            residential    industrial        highway     mixed
```

Fig. 11.1 Heirarchical Classification of Sites

11.1.1 Factors used in site classification

The factors used to classify sites were receiving stream quality, sewered catchment area, and catchment land use. Catchment area was selected in preference to DAR because its measurement in this study was more accurate, and because it is always available to planners of new drainage systems.

The presence of sewage or other contamination was not specifically included in the classification, since the results of data analysis were inconclusive (see section 9.5.3). It was also assumed that illegal connections would not be a consideration when systems were planned, although it is a problem which should be addressed separately. The effects of illegal discharges were indirectly taken into account in the analysis, as sites where contamination had been recorded were not excluded from the classification.

Receiving stream quality, which was identified in Chapter 9 as the most important factor influencing impact, was used at the first level of classification. Sites were placed in six quality classes, based on BMWP scores and corresponding to Severn-Trent Water's "General Biological Description" (section 7.2.1).

Catchment area and land use were used at the second and third levels of classification respectively, using the categories employed for the analysis described in Chapter 9.

11.1.2 Numbers of sites at different classification levels

The more detailed the classification, the fewer sites in each group, and the lower the level of confidence which could be placed in the results. Many classification groups were not represented by any sites, often because the combination of characteristics was unlikely, as in the case of highway drains serving areas over 100 ha. Other empty classification groups indicated site types which need further investigation, and these are identified below.

At the first level of classification, most sites were in the poor and moderate quality groups (Fig. 11.2). This obviously reflects the quality of urban streams in the Severn-Trent area, but more data are needed for outfalls to higher quality streams; particularly since new developments are often in semi-rural areas where stream quality is likely to be better than in established urban areas. Only one stream was in the unsatisfactory quality group, but as protection of such streams would not be a planning consideration, further data are not needed.

BIOLOGICAL QUALITY
OF RECEIVING STREAM

unsatisfactory	poor	moderate	good	very good	excellent
1 SITE	20 SITES	24 SITES	2 SITES	0 SITES	0 SITES

Fig. 11.2 Numbers of Sites in Biological Quality Groups

At the next level of classification, where sites were grouped according to sewered catchment area, the poor and moderate quality groups both included sites of all area classes (Fig. 11.3). The two good quality sites were a 12.5 ha residential estate and a 115 ha mixed land use site. More data are needed for catchments of all areas and land uses which drain to good quality streams.

POOR BIOLOGICAL QUALITY SITES — CATCHMENT AREA

<1ha	1-10	11-50	51-100	>100
5	7	4	3	1

MODERATE BIOLOGICAL QUALITY SITES — CATCHMENT AREA

<1ha	1-10	11-50	51-100	>100
3	11	7	1	2

Fig. 11.3 Numbers of Sites in Area Groups

Finally, the poor and moderate quality sites in each area class were grouped by catchment land use (Fig. 11.4). Most groups were represented by at least one site, with the exception of single-land-use groups for catchments over 50 ha; and it was assumed that this was because large catchments rarely have a single land use. Since there were so few sites in the 51-100 and >100 ha area groups, and no land use distinctions were to be made, these two area groups were subsequently merged. Other groups which were not represented were industrial areas under 1 ha, highway areas over 1 ha, and mixed land use catchments under 1 ha draining to moderate quality streams.

11.1.3 Biological impacts at different classification levels

A summary of biological impacts in terms of mean BMWP score differences is presented in Fig. 11.5, where sites are classified by stream quality and catchment area. As demonstrated in Chapter 9, the poor quality streams are less likely to be adversely

```
                    POOR BIOLOGICAL QUALITY SITES
                                 |
                                 |
                                 |
                           CATCHMENT AREA
                                 |
                                 |
      ---------------------------------------------------------
      |            |             |             |             |
      |            |             |             |             |
    <1 ha         1-10         11-50        51-100       >100 ha
      |            |             |             |             |
      |            |             |             |             |
   LAND USE     LAND USE      LAND USE      LAND USE      LAND USE
      |            |             |             |             |
   ___|___      ___|___       ___|___       ___|___       ___|___
   |  | |  |    |  | |  |     |  | |  |     |  | |  |     |  | |  |
   R  I H  M    R  I H  M     R  I H  M     R  I H  M     R  I H  M

   1  * 3  1    4  1 *  2     2  1 *  1     *  * *  3     *  * *  1


                  MODERATE BIOLOGICAL QUALITY SITES
                                 |
                                 |
                                 |
                           CATCHMENT AREA
                                 |
                                 |
      ---------------------------------------------------------
      |            |             |             |             |
      |            |             |             |             |
    <1 ha         1-10         11-50        51-100       >100 ha
      |            |             |             |             |
      |            |             |             |             |
   LAND USE     LAND USE      LAND USE      LAND USE      LAND USE
      |            |             |             |             |
   ___|___      ___|___       ___|___       ___|___       ___|___
   |  | |  |    |  | |  |     |  | |  |     |  | |  |     |  | |  |
   R  I H  M    R  I H  M     R  I H  M     R  I H  M     R  I H  M

   2  * 1  *    7  1 *  3     3  1 *  3     *  * *  1     *  * *  2
```

R = residential; I = industrial; H = highway; M = mixed
* = no sites in group

Fig. 11.4 Numbers of Sites in Land Use Groups

BIOLOGICAL QUALITY OF RECEIVING STREAM

Stream quality	Catchment area	n	[n]	(n %)
unsatisfactory		1	[+10]	(0 %)
	<1ha	*	*	*
	1-10	*	*	*
	11-50	1	[+10]	(0 %)
	>50	*	*	*
poor		20	[+1]	(45 %)
	<1ha	5	[0]	(60 %)
	1-10	7	[+6]	(0 %)
	11-50	4	[-2]	(50 %)
	>50	4	[-5]	(100%)
moderate		24	[-6]	(83 %)
	<1ha	3	[-6]	(100%)
	1-10	11	[-5]	(82 %)
	11-50	7	[-7]	(71 %)
	>50	3	[-5]	(100%)
good		2	[-6]	(50 %)
	<1ha	*	*	*
	1-10	*	*	*
	11-50	1	[+8]	(0 %)
	>50	1	[-19]	(100%)

n = number of sites in group
[n] = group mean BMWP score difference
(n %) = proportion of sites in group with reduced downstream BMWP score

Fig. 11.5 Biological Impacts for Sites Classified by Stream Quality and Sewered Catchment Area affected by discharges than the moderate quality streams: 45% of poor quality streams had reduced BMWP scores downstream of the outfall, compared with 83% of moderate quality streams. Since only two good quality streams were sampled, the 50% score reduction rate quoted in Fig. 11.5 is not a reliable estimate of the true proportion of good quality sites which are adversely affected by outfalls.

In all cases where the sewered catchment area was over 50 ha, there was a reduction in biological quality downstream of the outfall, with the greatest impact at the single site where a large catchment drained to a good quality stream. For smaller catchment areas, the observed impacts were more variable. Neither the proportion of adversely affected sites nor the magnitude of the impact increased with catchment area within the poor and moderate quality groups; and a high proportion of streams receiving discharges from catchments smaller than 1 ha had reduced BMWP scores downstream of the outfall.

These variable impacts at poor and moderate quality sites with catchment areas up to 50 ha were partly explained by further classification of sites, to include land use (Fig. 11.6), although the small groups at this level of classification did not allow any significance to be attached to the observations.

With the exception of one mixed land use catchment, 40% of which was highway, the BMWP score reductions at poor quality sites under 11 ha were for highway drains. The only highway drain to a moderate quality stream also resulted in a lower BMWP score downstream of the outfall. Highway drains, however small, and whatever the quality of the receiving stream, appeared to have a detrimental effect on water quality.

None of the residential catchments draining to poor quality streams adversely affected water quality, but those draining to moderate quality streams generally resulted in lower BMWP scores

```
                    POOR BIOLOGICAL QUALITY SITES
                                  |
          ------------------------|------------------------
          |                       |                       |
      CATCHMENT               CATCHMENT               CATCHMENT
        AREA                    AREA                    AREA
          |                       |                       |
        <1ha                    1-10                    11-50
          |                       |                       |
   -------|-------         -------|-------         -------|-------
   |    |     |    |       |    |     |    |       |    |     |    |
   R    I     H    M       R    I     H    M       R    I     H    M

   1    *     3    1       4    1     *    2       2    1     *    1
[+16]   *   [-5] [-4]    [+8] [+2]    *  [+7]    [+6] [-11]   * [-8]
  0%    *    67% 100%     0%   0%     *   0%      0%  100%    * 100%


                  MODERATE BIOLOGICAL QUALITY SITES
                                  |
          ------------------------|------------------------
          |                       |                       |
      CATCHMENT               CATCHMENT               CATCHMENT
        AREA                    AREA                    AREA
          |                       |                       |
        <1ha                    1-10                    11-50
          |                       |                       |
   -------|-------         -------|-------         -------|-------
   |    |     |    |       |    |     |    |       |    |     |    |
   R    I     H    M       R    I     H    M       R    I     H    M

   2    *     1    *       7    1     *    3       3    1     *    3
 [-6]   *   [-7]   *     [-12] [-8]   *  [+9]    [-4] [-16]   * [-6]
 100%   *   100%   *      86% 100%    *   67%     33% 100%    * 100%
```

R = residential; I = industrial; H = highway; M = mixed
n = number of sites in group; * = no sites
[n] = group mean BMWP score difference
n% = proportion of sites in group with reduced downstream BMWP score

Fig. 11.6 Biological Impacts for Different Land Uses regardless of catchment area. There was no evidence that the magnitude of the impact increased with catchment size.

Industrial catchments were poorly represented, but both those draining to moderate quality streams had an adverse effect, and the larger of the two which drained to a poor quality site also resulted in a fall in biological quality. More data are needed on industrial catchment outfalls, which are potentially damaging to water quality.

Mixed catchments, which generally included highway or industrial areas, resulted in a fall in BMWP score at five out of six moderate quality sites, but only two out of four poor quality sites. As for industrial catchments, more data are needed for mixed land use areas, possibly with further classification to indicate the nature of the land use mix.

11.2 TENTATIVE GUIDELINES FOR THE ESTIMATION OF DISCHARGE IMPACTS

Tentative guidelines, based on the classification of sites described above, have been produced so that the likely water quality effects of surface water sewer outfalls can be considered when drainage systems are planned.

The guidelines are in the form of a decision tree (Fig 11.7), derived from the data presented in Figs. 11.5 and 11.6. Working systematically down the tree by answering questions on catchment characteristics, the user is led to either a triangle, indicating a likely adverse impact on water quality, or to a circle, indicating no adverse impact. Within each triangle and circle is the number of observations on which the impact/no impact decision is based, expressed as a fraction of the number of sites belonging to the group. Where an impact is likely, the group mean BMWP score difference is given at the base of the triangle.

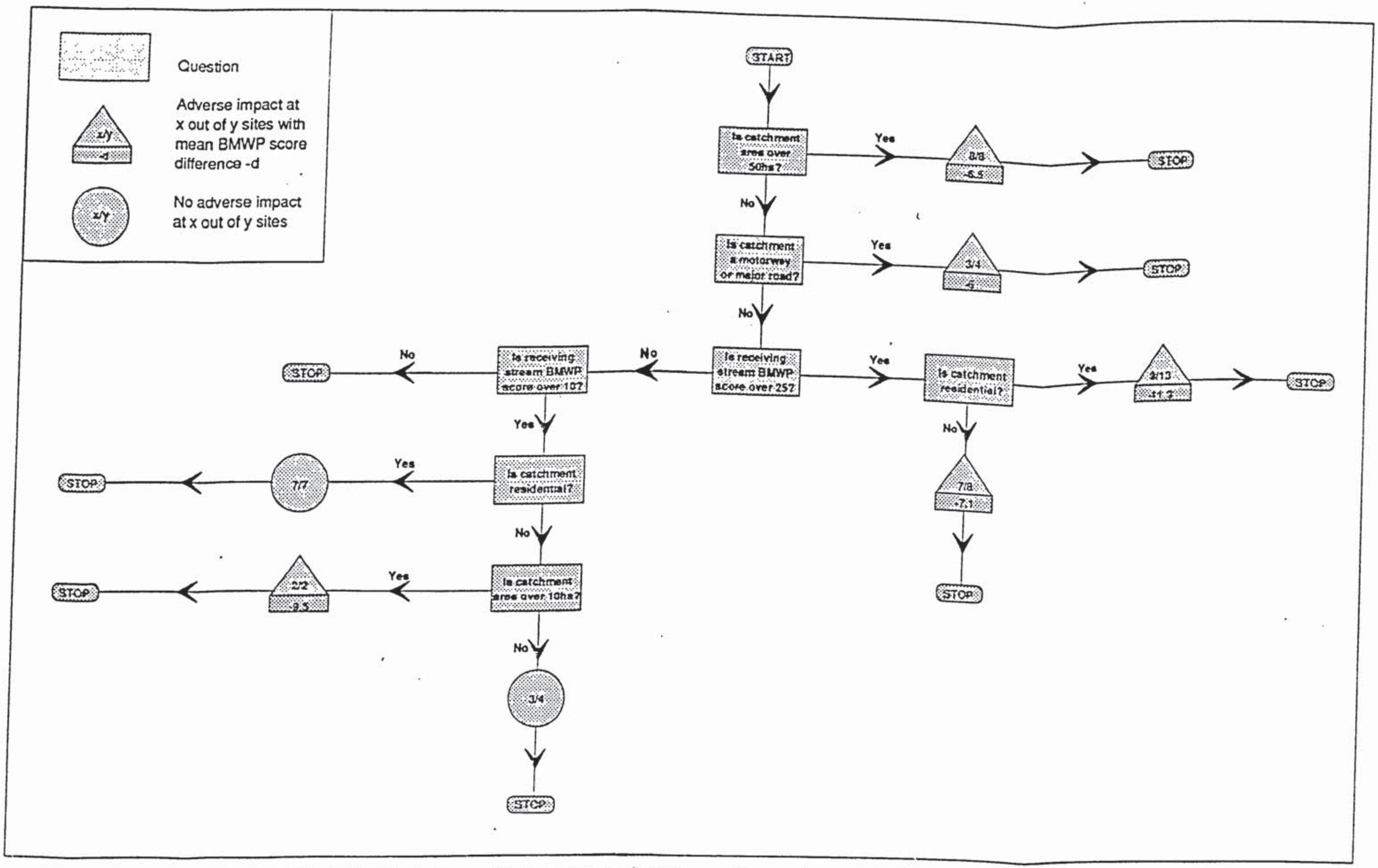


Fig 11.7 Tentative guidelines for estimating surface water sewer discharge impacts

For example, following the path for a proposed 12 ha residential estate draining to a stream with a BMWP score of 25, the user arrives at a circle (no impact) containing the fraction "7/7". This means that of 7 residential catchments studied which were under 50 ha and drained to poor quality streams, no adverse impacts were detected. Alternatively, if the same catchment drained to a stream with a BMWP score of 35, an adverse impact would be likely, since 9 out of 13 such sites studied had lower BMWP scores downstream of the outfall, with an average score reduction of 11.9.

The guidelines were derived by stepwise elimination of sites, using the minimum information necessary at each step. Catchments over 50 ha and highway catchments are thus eliminated at the top of the tree, since their impacts did not appear to be influenced by any other factors. For the remaining sites, receiving stream quality was the controlling factor, followed by land use. The tree branches were terminated when further questions did not result in higher values for the impact fractions.

The small number of sites studied and the lack of data for some site types, particularly catchments draining to good quality streams, does not allow complete confidence to be placed in the guidelines. Rather, it is intended that the tentative guidelines be taken as a starting point from which to build a more reliable decision network for impact estimation. Suggestions for further work which could be carried out to achieve this are given in section 11.3.

11.3 RECOMMENDATIONS FOR FURTHER WORK TO IMPROVE THE RELIABILITY OF IMPACT ESTIMATION

The major requirement of further work is the collection of more upstream-downstream biological data for various catchment types. There is also scope for improvement in the processing of biological data, the use of indices, and the classification of sites by catchment characteristics.

11.3.1 Collection of biological data

The justification for using BMWP scores in this project was twofold: BMWP scores are easy to derive because they only require identification of organisms to family level; and any impact not detected by BMWP score differences would not be of much interest at the screening level of impact assessment adopted here. Having demonstrated that BMWP score differences do reveal impacts, however, it is reasonable to attempt to refine impact definition by the use of other indices which take account of different factors. This would hopefully allow the magnitude of the impact to be quantified more reliably than by the use of BMWP scores, and could reveal additional, more subtle, effects. It may also be possible to explain the apparent improvements in stream quality observed at some sites, which has not been attempted in this study. Since BMWP scores are calculated from the minimum of biological data, they could always be used if it was found that alternative indices did not offer any advantages.

The use of weighted BMWP scores was shown in section 9.1.1 to reveal upstream-downstream differences which were statistically more significant than simple BMWP score differences. This indicated that organism abundance was affected by discharges, and in some instances this could be the only indication of a biological impact. It follows that any index used as an alternative to BMWP score for impact assessment should incorporate organism abundance. A relatively simple-to-derive index of this type is Chandler's Biotic Score (Chandler, 1970), which recognises different levels of abundance without counting of individual organisms. Hellawell (1986) rates Chandler's Score highly on the basis of its theoretical basis, ease of derivation, performance as a management tool, and subjective assessment of its performance. Other indices which could be investigated further at the species level are comparative measures which take account of abundance, although these would be more time-consuming to derive.

11.3.2 Catchment characteristics data

To improve the reliability of impact estimation, additional data are required for all site types, but particularly for those not covered by this study. The major gap in the present data set is for catchments draining to streams of good biological quality (BMWP score 50+). Other site types which merit special attention are non-residential catchments of all sizes, since there is evidence that these may be more damaging to water quality than residential estates.

It might also be possible to improve impact estimation by the use of a more sophisticated land use classification system. Mixed catchments, in particular, could be better defined; and it may be possible to identify the component land uses which are the cause of the impacts observed for mixed catchments. More detailed classification of residential areas might help to explain the variation in recorded impacts: the type of housing or the population density could be used as indicators of the general 'cleanliness' of the area, which has been quoted by Olivieri et al (1989) as a controlling factor in runoff quality. Industrial and highway catchments could also be sub-classified, as it may be that some light industrial sites do not pose a threat to water quality, or that only motorways have an adverse impact.

A final area in which data collection could be enhanced is for the receiving stream catchment area. Drainage area ratios have not been considered in the derivation of the tentative guidelines, but if more data were available it may be possible to separate the effects of sewered catchment area from DAR. This could permit impacts of large catchments, which in this study were always damaging to water quality, to be better assessed by consideration of the size of the receiving stream.

11.4 APPLICATION OF GUIDELINES FOR IMPACT ASSESSMENT

The proposed method of impact estimation does not take any account of either the acceptability of the predicted impact or alternative engineering solutions to the surface water drainage problem. Although these are beyond the scope of this project, the major points which should be considered are summarised below, with examples from the field survey.

11.4.1 Acceptability of predicted impacts

Whether or not an estimated impact is acceptable will depend on the designated use of the receiving stream. An estimated fall of 10 in BMWP score might be considered insignificant for many streams, but a stream supporting a healthy fishery could suffer as a result of discharges causing such an impact.

Many urban streams are not of sufficient quality to support fisheries, however, and recreation is a more common use. Three of the outfalls studied clearly impaired the recreational use of the receiving stream, and these are described briefly below.

As described in section 7.1.4, downstream of the outfall from the airport drain (Site 15, photographs (iii) and (iv)) there was excessive weed growth caused by nutrient enrichment of the water. This stream fed into a park boating lake, which had also become overgrown with weeds, to such an extent that it could no longer be used for boating.

The surface water from site 44 was conveyed for a short distance by a small brook before entering a culvert and discharging to the larger Smestow Brook. The small brook passed through a wildlife park, and had been stocked with frogs and other animals, but these were unable to survive the passage of the surface water, which was obviously polluted with metal finishing effluent from the industrial estate which the sewer system drained.

The third site at which recreational water use was seriously impaired was site 12 (photograph (v)) where there was a heavy dry weather sewage flow. This large outfall was to a stretch of the River Cole in Birmingham which flowed through a recreation ground, and the presence of sewage was aesthetically offensive as well as a potential threat to health. This example also illustrates the importance of public perception of pollution from surface water sewers, which although difficult to quantify can play a major part in recreational use impairment: during the course of the study complaints about discharges from the outfalls were common, even at sites where no adverse impact was detected.

At all three sites described above there was a reduction in BMWP score, of 16, 11, and 5 points for sites 15, 44 and 12 respectively; and in all cases the use of the tentative guidelines predicts an adverse impact on water quality.

11.4.2 Alternative engineering solutions

Once an adverse impact has been predicted and judged unacceptable, it becomes necessary to seek an alternative to the direct discharge of surface water to a stream. The alternatives fall into three broad categories:

a) Discharge to the combined system
b) Discharge to another watercourse, or to a different point on the same watercourse

c) Source control or runoff treatment to reduce the pollution threat

The advantages of discharge to the combined system must be set against the possible increased frequency of overflow operation. It may be that the risk of pollution from CSOs is far greater than that from the proposed surface water outfall, but in other cases an existing combined system may be able to take additional inputs without raising overflow frequency or volumes to an unacceptable level.

Discharge to another watercourse may be a viable alternative in cases where there is a larger stream close by which can accept the surface water sewer flows with a reduced threat to quality. It may also be an alternative where local water use is likely to be impaired by the presence of an outfall, in which case relocation of the discharge point may also be acceptable.

The management of urban runoff, and various strategies for reducing the pollution load on receiving streams were discussed in section 2.3. The cost of source control or treatment measures is likely to limit their use, but there are some relatively low-cost alternatives - such as the use of grassed swales for highway runoff - which are more attractive. There is a need for procedures to be developed which enable the relative merits of different management strategies to be assessed.

CHAPTER 12

CONCLUSIONS AND RECOMMENDATIONS

Pollution caused by discharges from urban drainage systems, both combined and separate, should be taken into consideration when systems are designed. To make this possible, and for the rational assessment of design alternatives, planning authorities require a means of predicting the water quality effects of such discharges.

Current research in the UK is directed towards the development of simulation models capable of analysing flows and pollutant loads in sewerage systems. These models will eventually be used by planners in conjunction with river quality models and standards designed to take account of the intermittent nature of storm discharges, but will first require data for calibration. There is a dearth of storm flow quality data for the UK, especially for separate surface water systems. Until such tools are operational, there is a need for procedures to estimate pollution loads and their effect on receiving water quality. Such procedures exist for combined sewer overflows, but there are no guidelines for separate surface water systems.

The presence of contaminants in urban runoff is well established, but their sources and quantities are less well understood. Studies in the UK, US and Europe have shown wide variation in contaminant levels in urban runoff, and have reached different conclusions on which factors control runoff quality.

Investigations of the impacts of discharges on receiving stream quality have mostly employed probablilistic methods, which rely on comparison of predicted in-stream pollutant concentrations with water quality standards from which environmental quality is inferred. A few studies have employed chemical and biological surveillance techniques, but these have been on the single catchment scale and have not been specific upstream-downstream studies to assess sewer discharge impacts. In the absence of a large national runoff and river quality database which would facilitate the development of probabilistic models, upstream-downstream studies such as the one described in the preceding chapters are a simple and direct means of assessing the impact on water quality of discharges from surface water sewers.

12.1 PRINCIPAL CONCLUSIONS

The four principal conclusions resulting from this study are as follows:

1. The field survey has shown that discharges from surface water sewers have a measurable impact on receiving stream quality.

2. The extent and nature of the impact varies from site to site. The overall effect is a deterioration in water quality which can impair the use of the watercourse downstream of the outfall.

3. Catchment characteristics influence impact and can be used to predict the likely impact of proposed outfalls from new developments.

4. Tentative guidelines for impact estimation have been produced for use by planning authorities. Further work should be carried out to extend these guidelines and improve their reliability.

12.2 DETAILED CONCLUSIONS

1. Overall, there was a small but significant fall in BMWP scores downstream of outfalls. The change in biological quality was also apparent in taxa counts and comparative indices, and became more significant when BMWP scores were weighted to take account of organism abundance.

2. There was no detectable impact on the levels of zinc, cadmium, lead, nickel, boron, chromium, vanadium, copper, silver, or arsenic in either sediment or algae samples.

3. Biological impacts were variable. Although in most cases the BMWP score downstream of the outfall was lower than the upstream score, there were instances in which water quality apparently improved.

4. One of the drawbacks to upstream-downstream studies is the lack of sites suitable for impact assessment. Sites must be chosen at which sampling stations differ only in their position relative to the outfall under investigation, and such sites are difficult to find. For this reason, it was not generally possible in this survey to sample further downstream than 100m from the outfall. At sites with an observed adverse impact where it was possible to sample further downstream, however, water quality did not have an opportunity to recover fully before another outfall discharged to the stream.

5. Overall, biological quality measured immediately downstream of the outfall and quality measured 100m downstream were both lower than upstream quality, but there was no consistent upstream-outfall-downstream pattern to the indices calculated. This could reflect different physical effects at the point of discharge.

6. Metals concentrations in algae generally reflected concentrations in sediments at the same station, but correlation between metals levels and biological quality was poor. Metals concentrations were apparently not high enough to affect BMWP scores.

7. Sewered catchment area influenced the biological impact of outfalls, with reductions in BMWP score more likely to occur downstream of outfalls from large catchments than downstream of small catchment outfalls. Catchment area had no effect on upstream-downstream differences in metals concentrations.

8. Catchment land use influenced biological impact. Industrial and highway catchments caused the greatest reduction in BMWP scores, and residential catchments had the lowest impacts. Land use had no effect on metals concentrations.

9. Many of the outfalls visited were operating in dry weather, often discharging foul sewage. Sewage was also present in some of the discharges observed at sites visited in wet weather. At least 28% of the sites studied had connections to the foul system. A further 4% were discharging industrial wastes in dry weather, and 10% were discharging unidentified wastes. The true extent of illegal connections and industrial malpractice could not be estimated since monitoring was not continuous, and it could be much greater.

10. Most outfalls with observed sewage discharges were damaging to water quality, but there were some instances in which quality improved downstream of sewage inputs.

11. The quality of the receiving stream was an important factor in determining outfall impact. At poor quality sites, where the BMWP score was 25 or below, the quality was as likely to improve as deteriorate downstream. At better quality sites, 80% suffered a reduction in BMWP score downstream of the outfall.

12. In several instances the presence of the outfall clearly impaired the recreational use of the receiving stream by affecting water quality. Public perception of outfalls may also affect recreational water use, even in cases where a fall in water quality can not be demonstrated using the methods employed in this study.

12.3 RECOMMENDATIONS

The principal recommendation of this project is that the tentative guidelines presented in Chapter 11 be extended and improved through further data collection, so that they can be used with confidence by planning authorities to estimate the water quality effects of discharges from proposed surface water sewer outfalls. This will allow damage to streams to be minimised by taking water quality into consideration when drainage systems are planned. In the meantime, the tentative guidelines can be used by planners for catchment types which were included in this study.

Future studies should pay particular attention to catchments discharging to clean streams, since data for these are sparse; and to highway and industrial catchments, which appear to be more damaging to water quality than residential areas. If suitable study sites can be found, the spatial extent of impacts should be investigated by sampling further downstream than was possible in this project. The cumulative effects of series of outfalls should also be considered, and alternative biological indices to BMWP scores should be investigated.

The second recommendation addresses the problems of illegal connections to the foul drainage system and disposal of household and industrial wastes via the surface water system. In many cases, the public are not aware that separate surface water drainage systems exist; and it is assumed that any wastes disposed of via household or roadside drains will receive treatment at sewage works. The same lack of knowledge can lead to illegal connections when domestic waste is directed into the nearest pipe. This situation could be rectified by the introduction of an information programme, designed to increase public knowledge of drainage systems, in which it is explained that separate sytems discharge directly into local watercourses.

The problem of deliberate use of surface water systems for industrial waste disposal is more difficult to deal with as it is often only detected once damage to water quality has occurred. The high impacts observed in this study at industrial catchment outfalls could be due to such illicit practices. Strict control of such practices would be difficult to achieve, and even if it was possible the risk of accidental spillages would still exist. Solutions should be sought to the drainage problems of areas where the risk of spillages or the potential for illegal disposal of wastes is high.

Finally, more thought should be given to the siting of outfalls. Many urban streams are used for recreation, and it has been observed that even in cases where water quality was not seriously affected, public perception of an outfall can spoil the recreational use of the receiving stream. In some instances this could be avoided by not siting large outfalls in parks, recreation grounds, nature reserves or other areas frequently used for recreation by the public.

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LIST OF ABBREVIATIONS

anova	Analysis of variance
BMWP	Biological Monitoring Working Party
BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
CSO	Combined sewer overflow
D	Kothe's Species Deficit
DAR	Drainage area ratio
DM	Modified Kothe's Species Deficit
EMC	Event mean concentration
ICP	Inductively coupled plasma
J	Jaccard's coefficient
LT	Less than
ND	Not detected
NGR	National (Ordnance Survey) grid reference
NS	Not sampled
NURP	Nationwide Urban Runoff Program (US)
PV	Permanganate value
TKN	Total Kjeldahl nitrogen
TON	Total oxidised nitrogen
TS	Total solids
TSS	Total suspended solids
WBMWP	Weighted BMWP Score